CHAPTER 4: RELATIONSHIPS THAT PROVIDE THE BASIS FOR DEFINING IMPACTS TO WATER RESOURCES

This chapter summarizes technical analyses conducted by the SFWMD Coastal Ecosystem Division to support the development of MFL criteria for Florida Bay. The MFL development process requires several steps, including the following: 1) identifying important resources and functions of Florida Bay, 2) surveying the available information and potential MFL approaches, 3) documenting historic conditions and developing a water budget, 4) determining technical relationships between freshwater inflow and salinity and determining these relationships' impacts on the bay's resources and functions and 5) developing numeric criteria that reflect the degree of impact that occurs to water resources as a function of freshwater inflow from the upstream watershed.

BACKGROUND AND SCOPE

Florida Bay is a shallow estuary (average depth <1 meter) dominated by a complex array of small islands and mud banks that restrict the internal circulation of water within the bay. Freshwater inputs into Florida Bay from its feeder watershed area in the Everglades occur largely in the form of overland flow through Taylor Slough, the C-111 Canal basin and numerous small creeks that transverse the mangrove-dominated Everglades–Florida Bay transition zone before reaching the coastal embayments within northeastern Florida Bay. A significant volume of water from the Everglades also flows through Shark Slough to the Gulf of Mexico through Whitewater Bay, which is on the southwest coast of Florida, near the western boundary of Florida Bay, but the present analysis only considers the flows entering northeastern Florida Bay through Taylor Slough and the C-111 Canal system (**Figure 21** and **Figure 22**).

Within the rich history of research and monitoring activities in Florida Bay, no comprehensive analysis of information directly linking the responses of Florida Bay biota to changes in freshwater inflow and/or salinity had yet been compiled as of the outset of the present work effort. Following initial analyses of available information, several studies were carried out specifically to support the present MFL analysis, including various modeling efforts, most notably 1) a mass-balance hydrologic model, 2) a dynamic seagrass model and 3) statistical higher-trophic-level species models. Further model development and refinement will proceed over the next several years in support of CERP's Florida Bay and Florida Keys Feasibility Study, which is evaluating the restoration needs of Florida Bay. These modeling efforts will also provide greater predictive capability for future MFL evaluations. This chapter supports the initial development of MFL criteria for Florida Bay. Its objectives are the following:

- To describe data and methods considered for use in MFL development.

- To analyze hydrology and salinity conditions.

- To analyze specific ecologic consequences of a range of different hydrologic and salinity conditions within the Florida Bay ecosystem.
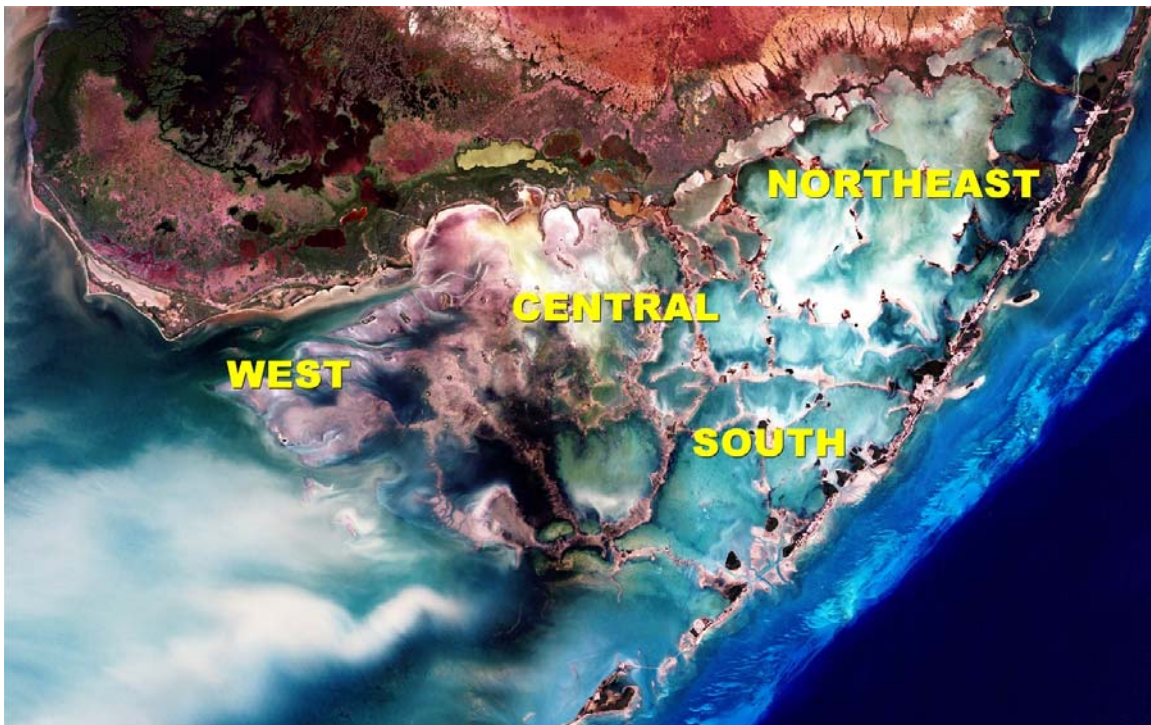
Figure 21. LANDSTAT-7 Extended Thematic Mapper Image of Florida Bay, Showing the Shallow Bank Bathymetry and Four Principal Subregions (from Florida Bay Science Program 2003).

Figure 22. Location of Gauged Inflow to Northeastern and Central Florida Bay (Hittle et al. 2001).

TECHNICAL METHODS AND DATA USED TO DEVELOP FLOW AND WATER LEVEL RELATIONSHIPS FOR THE MFL

Methods Considered

The methods used to determine water level and flow criteria were reviewed and categorized by Alber (2002) within the framework of the following three main areas of effects studied: 1) freshwater inflow effects, 2) estuarine-condition effects and 3) estuarine-resources effects. Freshwater inflow methods consider effects on the estuary that are related directly to quantity, quality or timing of inflow. Estuarine-condition methods contemplate effects on the estuary that are related to inflow characteristics of salinity, sediment or dissolved or particulate material. Estuarine-resources methods examine effects on the estuary related to organism/species composition, abundance, distribution or production in the inflow-affected area.

Within these three broad categories of effects, several possible approaches or methodologies can be considered for use in establishing water level and flow criteria. The following categories of approaches were recently summarized during development of the MFL criteria for the Northwest Fork of the Loxahatchee River (SFWMD 2002):

Instream Flow: There exist at least three general instream flow methodologies: 1) historic-flow techniques rely solely on preexisting data, 2) hydraulic techniques generally relate flow to the hydraulic geometry of a channel and 3) habitat methods relate flow to habitat suitability curves. When applied to estuaries, instream flow methods assume that the flow requirements of tributaries are commensurate with the flow requirements of the estuary. These methods are considered <u>freshwater inflow</u> approaches.

Hydrologic Variability: The hydrologic-variability approach extends instream flow techniques to include a more extensive analysis of flow characteristics. This approach also assumes that the freshwater needs of tributaries are the same as, or commensurate with, those of the estuary. An untested but feasible application of the method would be to use it with salinity data rather than flow data. This is also considered a <u>freshwater inflow</u> approach.

Habitat Overlap: As originally formulated, the habitat overlap approach has three steps: 1) salinities favorable for a particular species or group of species are identified, 2) the location in the estuary of favorable stationary habitat (such as sediment type or SAV) is determined and 3) freshwater inflows that create overlap between desired salinity and stationary habitat are identified. To date, dynamic habitat variables other than salinity have not been considered. This is considered an <u>estuarine-condition</u> approach.

Indicator Species: The indicator species approach relates a change in abundance, distribution or condition of a particular species to flow or salinity. Criteria for selection may include a species' endemism to the locale, its status as a species at risk, its ecologic importance and/or its commercial, recreational or aesthetic value. Statistical methods can be applied as a means to match species abundance values or species condition to appropriately time-lagged inflow or salinity conditions. This is considered an <u>estuarine-resource</u> approach.

Valued Ecosystem Component: An extension of the indicator species approach, analysis based on valued ecosystem components (VEC analysis) accounts for more known or suspected intermediate variables. Recommended by the United States Environmental Protection Agency (1987) for national estuary programs to characterize constraints on living resources, VEC analysis plays an important part in a general model for the design of eutrophication monitoring programs in South Florida estuaries. VEC is a goal-driven approach that has the ability to focus

research and to provide managers with short-term alternatives in data-poor estuaries. This is considered as another <u>estuarine-resource</u> approach.

In developing an approach to establish water level and flow criteria in Florida Bay, several sources of information were reviewed, including the following: 1) freshwater flow management methods being used in riverine estuaries nationwide and elsewhere in Florida (Estevez 2000), 2) a special issue of the journal Estuaries dedicated to minimum flows (Estuarine Research Federation 2002), 3) other coastal/estuarine MFLs (Caloosahatchee, Loxahatchee and St. Lucie) established at the District and 4) published literature and reports specific to Florida Bay to help identify potential indicator species or VEC, as well as available sources of hydrologic, physical and historical information.

Proposed Approach

A resource-based approach using the submerged aquatic vegetation (SAV) indicator species *Ruppia maritima* (widgeon grass) in the Everglades–Florida Bay transition zone is proposed for Florida Bay. Impacts to this resource are defined in terms of a freshwater flow regime and corresponding salinity levels required for survival of this SAV habitat. Using a 33-year historical time period 1970–2000, which includes drought conditions and changes in water management in the basin, the inflow to northeastern Florida Bay is determined. During the periods characterizing impacts to resources in the transition zone, concurrent inflow and resulting salinity conditions in northeastern Florida Bay are considered. The inferred effects on the northeastern Florida Bay seagrass community and upper-trophic-level species are described under these low flow conditions to assess the impacts of a low flow on the downstream Florida Bay ecosystem.

A representative gradient traversing the Everglades–Florida Bay transition zone into northeastern Florida Bay is used. This gradient comprises the following three regions:

- Ponds in the Taylor River region of the mangrove-dominated transition zone.

- Little Madeira Bay (a coastal embayment on the northern boundary of Florida Bay).

- A northeastern Florida Bay open-water area (Eagle Key Basin).

The gradient includes SAV communities ranging from 1) freshwater SAV (dominated by *Ruppia maritima*) at the inland ecotone (transition area between two different ecological communities) to 2) mixed seagrasses that are dominated by *Halodule wrightii* (shoal grass) and *Thalassia testudinum* (turtle grass) in the coastal transition zone and Florida Bay (**Figure 23**). The gradient is located in the part of Florida Bay that receives most of the fresh water that flows directly into the bay from the Everglades, and salinity along this gradient is influenced by water management. This gradient is appropriate for several reasons, including the following:

- The gradient originates in ponds within the Taylor River region's upland ecotone, which represent an environment that typically supports predominately freshwater to brackish-water biota on an annual basis and is highly sensitive to saltwater intrusions.

- The gradient passes through downstream areas that include a representative coastal embayment (Little Madeira Bay)that receives direct freshwater inflow from Taylor Slough. The environment and salinity regime are similar to those of other coastal embayments receiving freshwater inflow, such as Long Sound or Joe Bay.

Figure 23. Map of the Everglades–Florida Bay Transition Zone and Northeastern Florida Bay, Showing Gradient Location (the gradient is denoted as { }, with key monitoring stations shown). Red rectangle north of Little Madeira Bay is the area of the image in **Figure 32**.

- The endpoint is a well-mixed location within northeastern Florida Bay (Eagle key Basin) that is similar to most of the rest of eastern Florida Bay. Salinity and biota along this transect respond to freshwater inflows from creeks and sheet flow along the northeast Florida Bay coast.

- Monitoring of flow, salinity and SAV has been ongoing at several locations along this transect. 1) Flows are monitored at the mouth of the Taylor River by USGS. 2) Salinity is continuously monitored at an upstream Taylor River site (TR), at Taylor River mouth (TM) in northern Little Madeira Bay and outside of the mouth of Little Madeira Bay (LM) in northern Eagle Key Basin; salinity is also monitored monthly at the LM site and several other northeastern Florida Bay locations. 3) SAV species have been monitored seasonally in Little Madeira Bay and Eagle Key Basin (Fourqurean et al. 2002) and in the transition zone by the National Audubon Society.

- A multiple linear regression (MLR) model for Taylor River provides reasonable estimates of salinity at the TR site.

- A hydrologic model (FATHOM) for Florida Bay allows robust predictions of salinity along this gradient in Little Madeira Bay and for the adjoining northeast interior bay region's Eagle Key Basin, accounting for >75 percent monthly salinity variability.

- The transect encompasses a protected sanctuary for the American crocodile, a federally listed endangered species that requires access to fresh water.

The SAV community along this gradient is a critical component of the ecosystem. The presence of SAV species is required for key ecologic functions in the Florida Bay estuarine ecosystem, such as cycling of nutrients, provision of habitat for a range of species, provision of feeding grounds for waterfowl and stabilization of sediment. The presence of an estuarine condition that ranges from low to high salinities is an important feature for maintaining a diverse SAV community—including widgeon grass (*Ruppia maritima*), shoal grass (*Halodule wrightii*) and turtle grass (*Thalassia testudinum*) —that provides plentiful high-quality habitat and is able to support resident biota with needed shelter, food, good substrate and satisfactory water quality through sediment stabilization. The range of salinities is also important for fish and invertebrates that rely on the presence of estuarine conditions for all or part of their life cycle. Model analyses indicate the sensitivity of various fauna to salinity and to habitat quality, which itself is sensitive to salinity.

The technical information that will provide a basis to develop water level and flow recommendations for Florida Bay is presented in this chapter. The information includes historical measurements of flow from structures, water budget descriptions, laboratory mesocosm work on SAV growth and reproduction, field data and observations, literature review and modeling applications.

The modeling synthesizes past and present hydrology to allow a historical reconstruction of inflows and corresponding ecologic effects for Florida Bay and the Everglades–Florida Bay transition zone. This period of record is significant because it includes several periods of low flow resulting from drought conditions, as well as low flow periods resulting from water management activities. Statistical analysis provides evaluation of the transition zone SAV at varying salinities. Seagrass modeling provides evaluation of the Florida Bay seagrass community over the historical period. Statistical modeling of upper-trophic-level species and forage fish assemblage allows for the evaluation of the combined effects of changing salinity and SAV habitat. The following is a brief overview of the modeling approach:

- Hydrologic models were employed to develop a water budget and to predict surface water flows and salinity response leading from the Everglades–Florida Bay transition zone downstream into Florida Bay.

- The ecologic effects of salinity levels were evaluated along a gradient representing three areas of Florida Bay: 1) the Everglades–Florida Bay transition zone, 2) the northeastern coastal embayment area and 3) the open-water area of northeastern Florida Bay. This gradient was used because a relationship between inflow and salinity could be established. The evaluation included reviews of literature, statistical analysis of local monitoring data, analysis of experimental results and the development and application of ecologic models.

- *Ruppia maritima* was selected as an indicator species for the Everglades–Florida Bay transition zone. When salinity conditions are too high to allow Ruppia maritima survival in the transition zone, loss of the existing (predominately fresh water) SAV community is also expected to occur.

- A link between the ecologic health of *Ruppia maritima* in the transition zone and concurrent effects on Florida Bay seagrass communities is presented. When high salinity events cause the loss of *Ruppia maritima* within the Everglades–Florida Bay transition zone, negative ecologic impacts to northeastern Florida Bay can also be inferred, based on ecologic modeling of the SAV and higher-trophic-level species.

- Based on these data, a relationship between freshwater inflow and resource impact was developed for Florida Bay as discussed in Chapter 5.

HYDROLOGIC ANALYSES

Several hydrologic analyses were conducted to support development of flow-salinity relationships for Florida Bay. The present section summarizes 1) application of the mass-balance model FATHOM (Flux-Accounting Tidal Hydrology Ocean Model) to reconstruct a history of estimated salinities within 41 basins located in Florida Bay for period from 1970 through 2002 (see ECT, Inc. 2005) and 2) use of a multivariate linear regression model (MLR) to predict salinity at a station within the Everglades–Florida Bay transition zone (see Marshall et al. 2004).

Mass-Balance Hydrologic Model (FATHOM)

To assist in the development of water level and flow relationship for Florida Bay MFL development, the FATHOM model (ECT, Inc. 2005) was updated to represent 1) freshwater inflows from the upstream wetland and 2) salinity conditions in Florida Bay. The FATHOM model calculates variation in salinity in Florida Bay based on a mass-balance approach. Hydrologic inputs include monthly values of evaporation, sea level, boundary salinity, runoff, rainfall and tides at the boundaries of the model domain, updated to include spatially distributed rainfall and tides and direct measurements of freshwater runoff. Additional refinements were made including compilation of updated bathymetry, inflows and hydrologic data sets, as well as the use of time-varying salinity boundary conditions along the western boundary with the Gulf of Mexico. These updates reflect a significant improvement in detail and reliability of data inputs relative to the previously published description of FATHOM applied to Florida Bay (Cosby et al. 1999).

FATHOM is used to provide quantitative estimates of physical properties (such as basin residence times and salinity) on a monthly time scale under different hydrologic and flow scenarios. A historical reconstruction, spanning the period from 1970 to 2002, was developed to provide historical salinity estimates and annual water budgets for the 41 basins in Florida Bay. A water budget was constructed because this period comprises a wide range of climatic and inflow variations. Data that define the historical reconstruction period include estimated monthly rainfall, evaporation and freshwater inflow to the bay from the mangrove transition zone. The "base case" salinity predictions were based on calibration analyses from 1991 to 2002, a period with a comprehensive set of observed hydrologic data (ECT, Inc. 2005).

Water Budget

The Everglades–Florida Bay transition zone is an area of extensive mangrove wetlands consisting of shallow swamplands, creeks, ponds and bays along the mainland shore of northern Florida Bay. The major source of fresh water into Florida Bay traversing this ecotone is flow from the Taylor River and a series of approximately 20 creeks carrying surface water from the Taylor Slough/C-111 drainage area into the bay. The much larger Shark Slough basin, which under most conditions is hydrologically separate from the Taylor Slough/C-111 basin, drains into the Gulf of Mexico and is not considered in this study. Direct measurements of freshwater inflow into Florida Bay have been made since 1996 by the U.S. Geological Survey from five gauged creeks discharging into the Long Sound, Joe Bay, Little Madeira Bay and Terrapin Bay coastal embayments. Evidence from natural- tracers suggests that submarine groundwater discharge into Florida Bay contributes only slightly to the net freshwater supply (Corbett et al. 1999); therefore, this component is not included in the water budget. Ungauged flow has been estimated by the USGS in four additional creeks as constituting roughly an additional 23 percent of the gauged inflow (Hittle et al. 2001). Except for these empirical relationships, there appears to be no other information on the magnitude of the ungauged discharge of fresh water from the Everglades directly into Florida Bay. In any case, ungauged surface flow and ungauged groundwater are

expected to be greater in periods of high inflow rather than during the low inflow periods that are the focus of the present evaluation.

Fresh water first flows through extensive mangrove wetlands consisting of shallow swamplands, creeks, ponds and bays before reaching open portions of northeastern and coastal central Florida Bay. Florida Bay's watershed within the southern Everglades can be subdivided into three regions: 1) Long Pine basin, 2) Taylor Slough and 3) the C-111 basin (**Figure 24**). Discharge from Long Pine basin is the result of rainfall in excess of evaporation within the basin—there is no large surface inflow to this basin. Flow through McCormick Creek, the only gauged surface outflow from the Long Pine basin, occurs intermittently (ECT, Inc. 2005). Freshwater flow from Taylor Slough is a function of rainfall, evaporation and management of the L-31 Canal and associated structures at the head of the slough. Flow from the Taylor Slough subregion discharges into the bay via many small creeks, including Taylor River (which is the largest of these creeks.) Most of the water that flows from the C-111 Canal basin into the bay first travels from the canal into mangrove wetlands, then through many small creeks into Joe Bay or Long Sound and then into northeastern Florida Bay. During periods of relatively high flow, the S-197 structure located at the terminus of the C-111 Canal is opened and water discharges into Manatee Bay, which is part of the Biscayne Bay system.
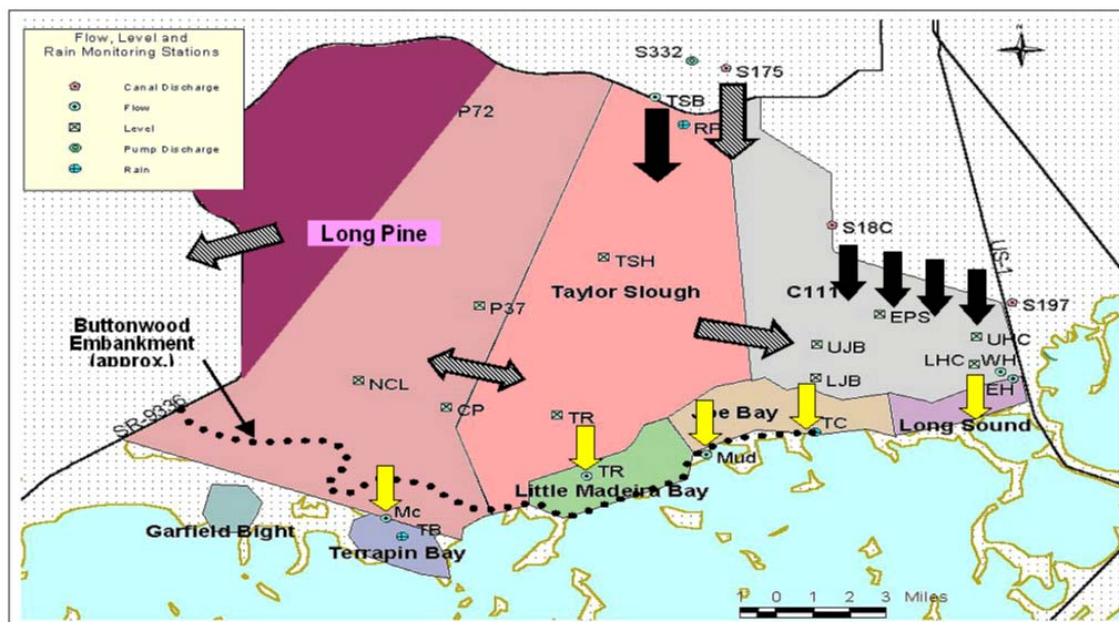

Figure 24.   Wetland Basins Used in the Hydrologic Analyses. Arrows show surface inflows to the wetland basins included in the water balance calculations [black], location of USGS measured creek flows (yellow) and calculated surface fluxes (hatched).

Long-term records (since 1970 for the canal control structures) of freshwater inflow to the southern Everglades include the records of flow at Taylor Slough bridge (TSB), which lies within Taylor Slough, and at canal structures S-175, S-18C and S-197 (**Figure 24**). The TSB and S-175 flows are the principal sources of surface water inflow into the Taylor Slough wetland basin. The S-18C flow minus the S-197 flow provides the basis to estimate overland discharge from the C-111 Canal into the downstream C-111 wetland basin and ultimately into the northeast corner of Florida Bay. The input to the FATHOM historical reconstruction includes some additional flow added to the measured flow at structures to account for excess rainfall over the wetland and for ungauged flow as detailed by ECT, Inc. (2005). Total average annual inflows to northeast Florida Bay from these sources show an increasing trend for the 31-year period 1970–2000 (**Figure 25**).
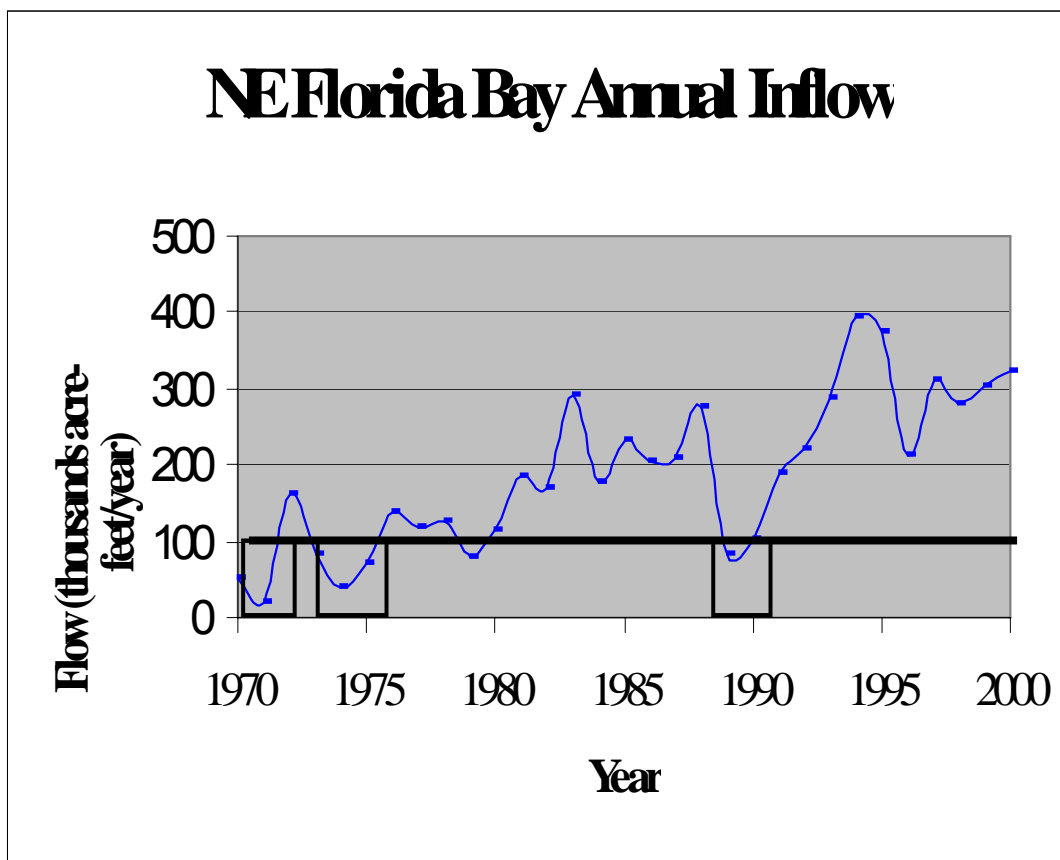
Figure 25.　Annual Overland Inflow to Northeastern Florida Bay, 1970–2000 . This information is an input to the FATHOM model and is based on measured structure flows (Taylor Slough bridge + S18C – S197); additional flow was added to the measured structure flow in the FATHOM model to account for excess rainfall over the wetland and ungauged flow as detailed by ECT, Inc (2005); boxed areas correspond to periods in which annual inflows (indicated by the symbols) fall below 105,000 acre feet/year for more than two consecutive years.

Sensitivity to Rainfall Variations

Patterns and distributions of rainfall were examined for the Everglades watershed and Florida Bay to determine the amount of rain typical of dry, normal and wet years (**Figure 26**). Rainfall data for the 31-year period 1970–2000 were ranked separately for the Everglades and Florida Bay. Rainfall analyses representing the Everglades (Shark River Slough and the Water Conservation Areas) were used from the Florida Climate Division 5 rainfall records (ECT, Inc. 2005); Division 5 records include data from numerous gauges within south Florida as compiled by the National Climatic Data Center of the National Oceanographic and Atmospheric Administration (NOAA). Florida Bay rainfall analyses were based on the spatially variable rainfall data from three stations bordering Florida Bay (Flamingo, Tavernier and Royal Palm) as described by ECT, Inc. (2005). In order to aid in the water budget interpretation, dry, normal and wet years were selected as years ranking near the 10 percent (dry), 50 percent (normal) and 90 percent (wet) thresholds of the annual rainfall distribution over the 31-year period.

Water Year	Fl. Bay Rainfall	Water Year	Div. 5 Rainfall		
1989	31.7	1976	41.4		
1971	34.7	1971	42.72		
1986	35.7	1985	44.02		Dry
1974	36.2	2000	44.45		
1985	40.9	1990	44.71		
1990	41.4	1981	44.81		
2000	43.4	1977	45.55		
1980	43.5	1972	46.53		
1984	43.6	1989	46.55		
1977	45.4	1988	46.91		
1970	45.9	1980	47.03		
1994	46.3	1996	47.38		
1979	49.2	1974	47.85		
1992	49.5	1975	48.62		Normal
1987	49.6	1997	48.97		
1996	50.4	1987	49.78		
1975	51.4	1986	50.5		
1991	52.4	1984	50.91		
1976	54.0	1973	51.09		
1993	54.9	1992	52.31		
1998	55.0	1994	53.6		
1978	55.6	1979	53.69		
1972	55.7	1993	55.15		
1997	57.5	1978	55.67		
1973	60.8	1970	57.46		
1982	61.3	1991	57.53		
1988	63.6	1998	57.56		
1981	63.8	1983	59.8		
1999	67.3	1982	60.41		Wet
1983	71.7	1999	63.22		
1995	72.4	1995	78.2		

Normal, Wet and Dry Years (by water year)
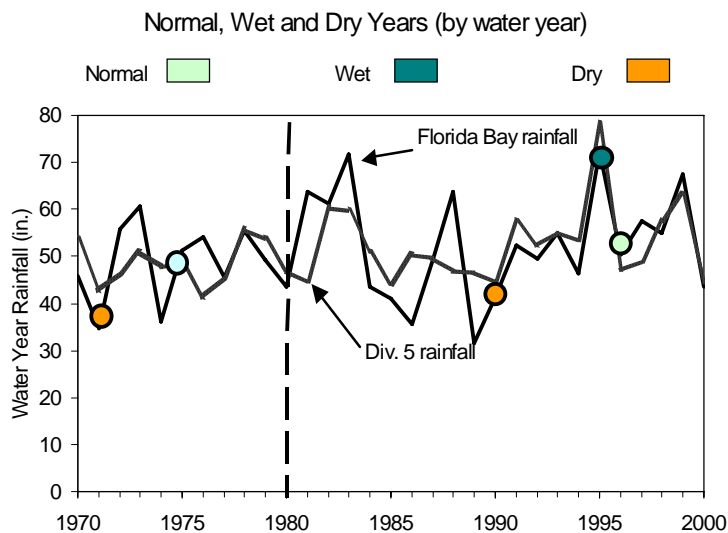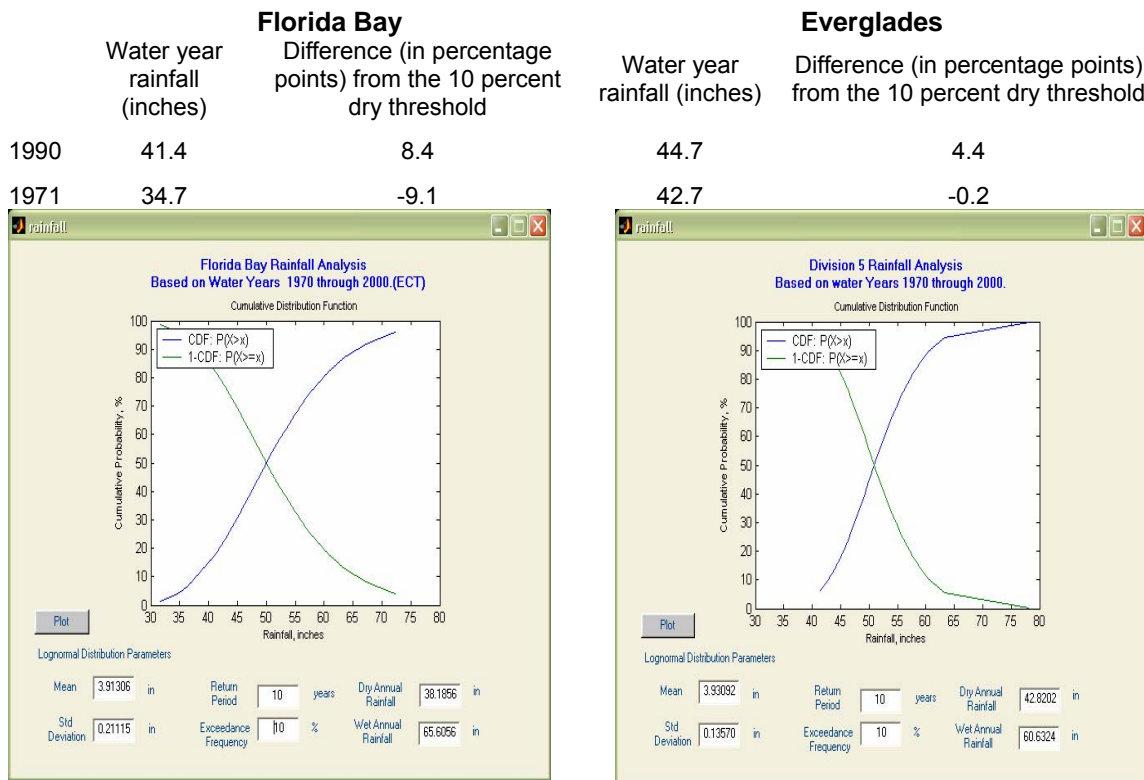
Normal   Wet   Dry

Figure 26. Ranking of Normal, Wet and Dry Years . This analysis is based on Florida Bay rainfall [used as FATHOM input] and Everglades rainfall [Div. 5] for 1970–2000; the water year is defined as November 1–October 31; rainfall patterns spatially differ, resulting in marked differences in rainfall amounts between Florida Bay and the southern Everglades [Div. 5] during some years; these years were excluded from the selection of representative wet, normal and dry years (ECT, Inc. 2005).

The 31-year record shows two representative drought years (November 1–October 31) near the 10 percent dry threshold level: 1971 and 1990 (**Table 9**). The two years were considered representative of a 1-in-10 year drought condition (defined as the 10 percent threshold having a return period of 10 years, thus occurring once every ten years on average). The year preceding the 1990 drought (1989) had the lowest rainfall measured over Florida Bay during the 31-year period of record, while the Everglades (Division 5) annual rainfall was more moderate (Error! Reference source not found.). Thus, Florida Bay felt the effects of near 1-in-10 year drought conditions for two consecutive years. The pre-1980 period shown was representative of conditions that were drier than normal (**Figure 26**) and does not contain a representative wet year (ECT, Inc. 2005).

Table 9. Representative Drought Years. The 10 percent threshold and deviation of the historical representative years were compiled using 1970–2000 data. The 10 percent threshold for the Everglades (Division 5) = 42.82 inches and for Florida Bay = 38.18 inches; these results are comparable to the 10 percent threshold calculated using data from the Flamingo (36.4", 40-year record) and Tavernier (30.8", 63-year record) rainfall stations.

	Florida Bay		**Everglades**	
	Water year rainfall (inches)	Difference (in percentage points) from the 10 percent dry threshold	Water year rainfall (inches)	Difference (in percentage points) from the 10 percent dry threshold
1990	41.4	8.4	44.7	4.4
1971	34.7	-9.1	42.7	-0.2

Sensitivity to Variations in Inflow from Canals

A notable factor affecting Florida Bay during the study period was that surface water discharges through Taylor Slough Bridge (TSB) and C-111 Canal were low in the period 1970–1981 relative to flows after 1981. As a result of changes in water management activities (**Table 2** and **Figure 13**) flows into the Everglades–Florida Bay transition zone increased by about a factor of four, relative to rainfall, after 1981. This is perhaps the most significant change that occurred in Florida Bay's freshwater budget during the period 1970 through 2000. To take this factor into account, normal and dry years were defined in both the pre-1980 and post-1980 periods, for comparison.

The period from 1973 to 1975, a period of low water delivery to the system, is highly variable, containing dry to wet years depending on year and location over Florida Bay or over the watershed. The water year 1975 was normal in terms of precipitation but had total annual inflow -- comparable to the 1989–1990 drought period (**Figure 27**). This illustrates effects of 1960–1980 regional water management activities that decreased flow to the mangrove transition zone.
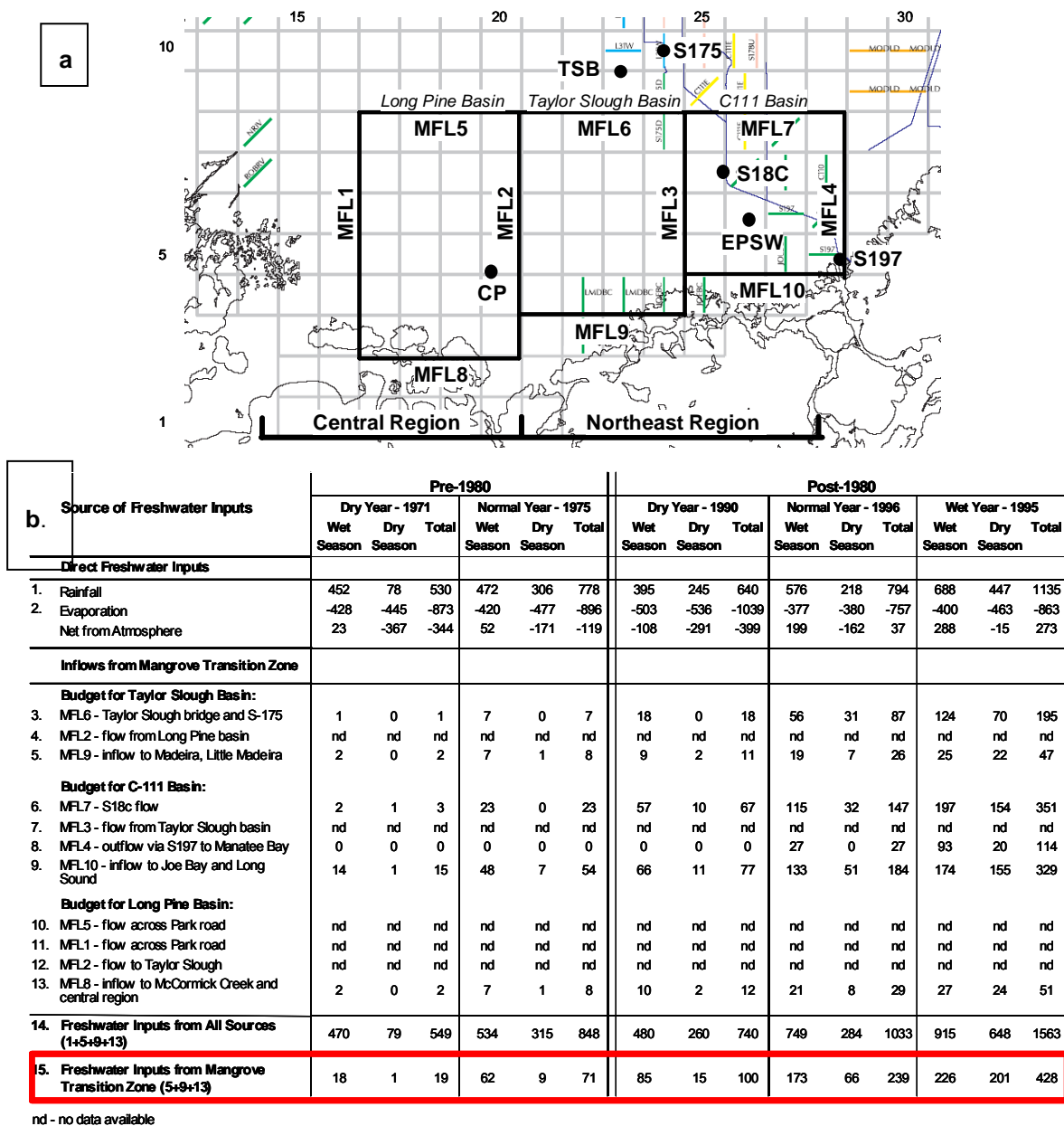
| b. | Source of Freshwater Inputs | Pre-1980 | | | | | | Post-1980 | | | | | | | | |
| | | Dry Year - 1971 | | | Normal Year - 1975 | | | Dry Year - 1990 | | | Normal Year - 1996 | | | Wet Year - 1995 | | |
		Wet Season	Dry Season	Total	Wet Season	Dry Season	Total	Wet Season	Dry Season	Total	Wet Season	Dry Season	Total	Wet Season	Dry Season	Total
	Direct Freshwater Inputs															
1.	Rainfall	452	78	530	472	306	778	395	245	640	576	218	794	688	447	1135
2.	Evaporation	-428	-445	-873	-420	-477	-896	-503	-536	-1039	-377	-380	-757	-400	-463	-863
	Net from Atmosphere	23	-367	-344	52	-171	-119	-108	-291	-399	199	-162	37	288	-15	273
	Inflows from Mangrove Transition Zone															
	Budget for Taylor Slough Basin:															
3.	MFL6 - Taylor Slough bridge and S-175	1	0	1	7	0	7	18	0	18	56	31	87	124	70	195
4.	MFL2 - flow from Long Pine basin	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
5.	MFL9 - inflow to Madeira, Little Madeira	2	0	2	7	1	8	9	2	11	19	7	26	25	22	47
	Budget for C-111 Basin:															
6.	MFL7 - S18c flow	2	1	3	23	0	23	57	10	67	115	32	147	197	154	351
7.	MFL3 - flow from Taylor Slough basin	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
8.	MFL4 - outflow via S197 to Manatee Bay	0	0	0	0	0	0	0	0	0	27	0	27	93	20	114
9.	MFL10 - inflow to Joe Bay and Long Sound	14	1	15	48	7	54	66	11	77	133	51	184	174	155	329
	Budget for Long Pine Basin:															
10.	MFL5 - flow across Park road	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
11.	MFL1 - flow across Park road	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
12.	MFL2 - flow to Taylor Slough	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
13.	MFL8 - inflow to McCormick Creek and central region	2	0	2	7	1	8	10	2	12	21	8	29	27	24	51
14.	**Freshwater Inputs from All Sources (1+5+9+13)**	470	79	549	534	315	848	480	260	740	749	284	1033	915	648	1563
15.	**Freshwater Inputs from Mangrove Transition Zone (5+9+13)**	18	1	19	62	9	71	85	15	100	173	66	239	226	201	428

nd - no data available

Figure 27. Simplified Water Budget for Northeast and Central Florida Bay: (a) Wetland Basins Used to Estimate Freshwater Inflow through the Everglades–Florida Bay Transition Zone (locations of the flow transects are superimposed on the grid of the SFWMM model; also shown are locations for flow [TSB, S175, S18C and S197] and wetland water levels [CP and EPSW]); (b) Flows Reported for Water Year November 1 1994 – October 31 1995 Pre- and Post-1980 (in 1000 ac-ft per year).

Effects of Inflow Changes upon Salinity

The average annual water budget for Florida Bay for the period 1970–2000 was compiled using the FATHOM base case (ECT, Inc. 2005). Rainfall and evaporation dominate the freshwater budget. On an annual basis, inflow is typically only about 20 percent of rainfall in the central and northeastern regions of the bay, but inflow's contribution is necessary to maintain a net positive inflow in late summer and fall (**Figure 28**).
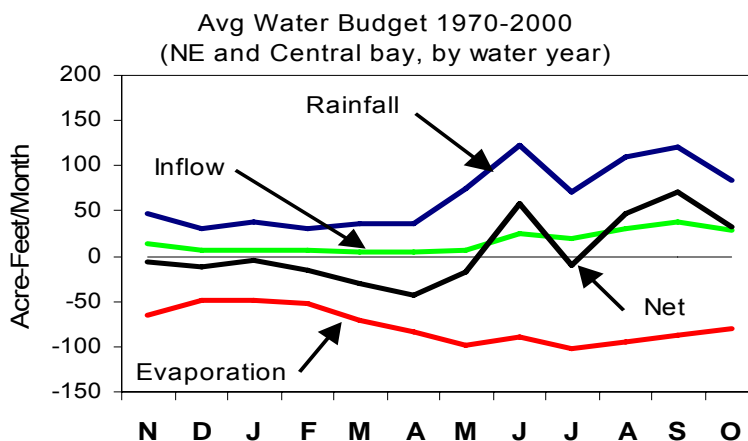

Figure 28. Average Water Budget, 1970–2000 . Evaporation and direct rainfall are the largest fluxes of fresh water into northeast and central Florida Bay; "net" refers to the difference between these terms; direct inflow is not overall a large component of the annual water budget but accounts for more than one-third of the net freshwater supply in late summer through the fall (ECT, Inc 2005).

A minimum flow specification will be ecologically relevant only to those parts of the bay that are influenced by inflow. Using results from the FATHOM hydrologic analyses, a linear statistical model was developed (ECT, Inc. 2005) to assess whether annual maximum salinity values within Florida Bay were sensitive to inflows and direct precipitation. Maximum annual salinity was indicated as being sensitive to inflow, rainfall or water level if the corresponding coefficient in the linear model tested significantly different from zero at the $p=0.05$ level. Bay basins in which annual maximum salinity is significantly correlated to year-to-year changes in inflow are clustered in the northeast and eastern interior (**Figure 29**).

Models for bay basins indicated that inflow changes did not explain the variation in maximum annual salinity in the west and western portion of the south region, presumably because maximum annual salinity values in those areas are a function of local rainfall and evaporation and salinity variation on the open western boundary of the bay. Western boundary conditions are primarily driven by changes in freshwater discharges from Shark Slough (which are not part of the flow analyses) and by oceanographic processes in the Gulf of Mexico.

In the central region (light blue area in **Figure 29**) the apparent lack of influence of inflow on maximum salinity may simply reflect the fact that very little or no inflow reaches the central region during dry years. In contrast, annual maximum salinities at select locations in the central bay are significantly correlated with wetland water levels (ECT, Inc. 2005). Given the small amounts of direct inflow to this region, however, additional analyses would be needed in order to quantify a relationship between water level and inflow in order for this finding to be useful for minimum flow determination. It is possible that much of the inflow that eventually enters the central bay first flows into the northeast bay and the complex mixing and circulation dynamics within the bay

determine the extent to which this freshwater influences the central bay. A quantitative estimate of this influence requires a hydrodynamic model, which is currently being developed as part of CERP's Florida Bay and Florida Keys Feasibility Study.
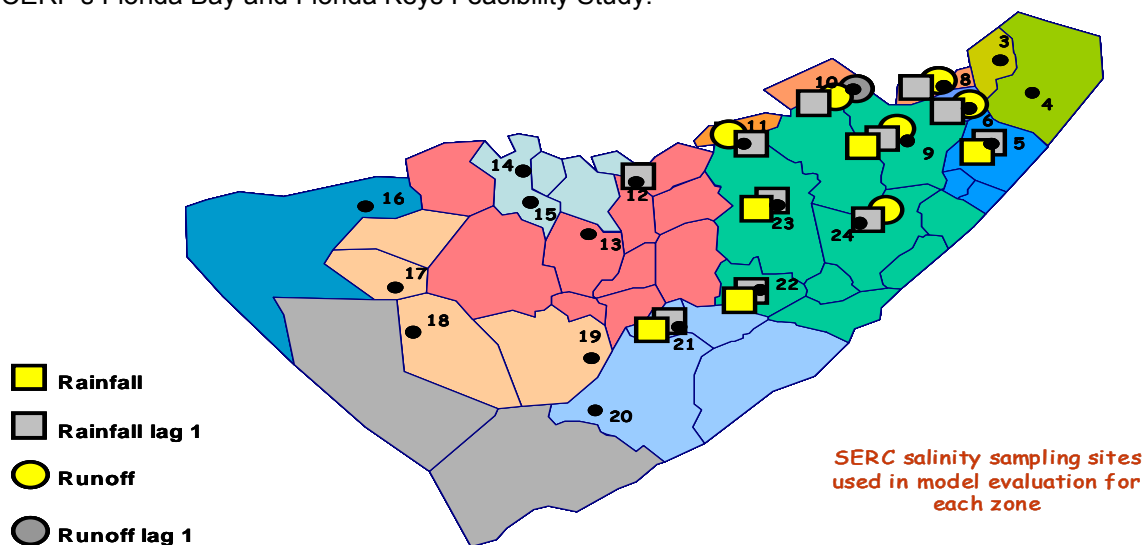

Figure 29. Regions of Inflow Sensitivity. The lag 1 analysis incorporates prior years' annual rainfall or runoff; shown are basins in Florida Bay where annual maximum salinity [calendar year basis] is significantly correlated to annual inflow in addition to annual rainfall; such areas are clustered in northeast Florida Bay; the regions are colored consistent with FATHOM analyses (ECT, Inc. 2005).

The FATHOM model was used to develop a 33-year monthly mean salinity time series for each of the 41 individual basins within Florida Bay (ECT, Inc. 2005). This base case represents the reconstruction of the water budget as close to historical conditions as possible. As illustrated by the calibration period (1991–2002), model fidelity and predictions varied somewhat by basin (**Figure 30**). Overall, the FATHOM model is capable of explaining about 81 percent of the monthly salinity variability throughout the 41 basins modeled within Florida Bay (ECT, Inc. 2005).

Performance of FATHOM varied from area to area. In general terms, the best performance was achieved in the northeast and eastern basins (shown as orange and blue-green basins in **Figure 30**. The lowest efficiency in these regions was in Joe Bay. Predictions in all regions do not reflect the monthly range of possible upper and lower daily extremes. It is important to recognize that the monthly mean predictions by FATHOM are compared against grab sample measurements taken during the month of comparison.
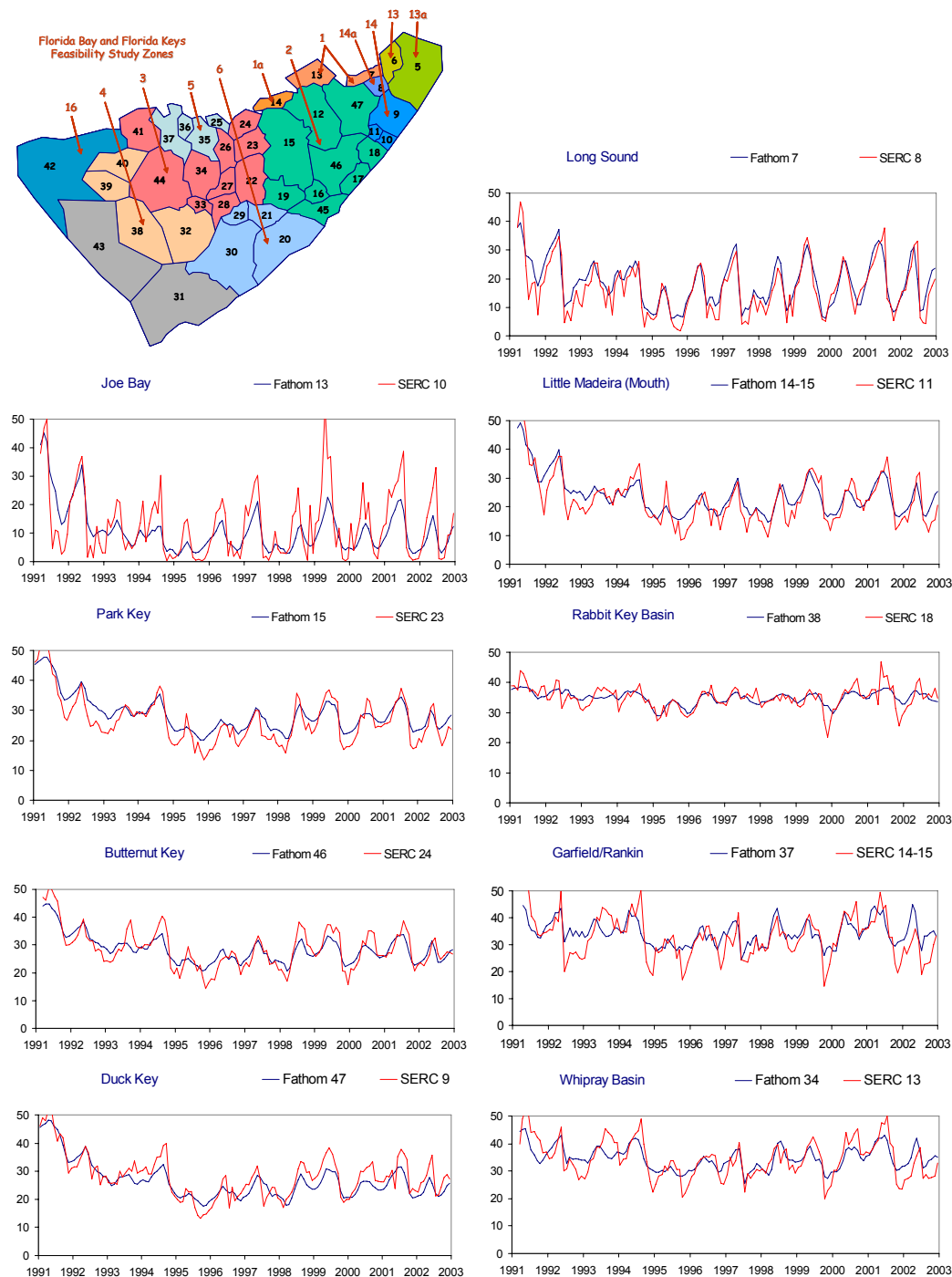
Figure 30. Time Series for Selected Florida Bay Basins, Showing FATHOM Predictions and Observed Values (SERC sampling stations) . The inset at the upper left shows the location of the FATHOM basins; in the present report, further analyses are shown for Little Madeira Bay (FATHOM basin 14) and for Park Key (FATHOM basin 15, referred to also as Eagle Key Basin) (ECT, Inc. 2005).

Everglades–Florida Bay Transition Zone Salinity Model

Analysis to determine water level and flow criteria for Florida Bay requires estimating salinity in the Everglades–Florida Bay transition zone (**Figure 23**). The FATHOM model estimates do not extend into that zone, and so a statistical modeling approach was used. The Taylor River station (TR) is located in this ecotone along the representative gradient and is part of the ENP Marine Monitoring Network; thus TR is used in the present analysis as an indicator site for the transition zone. During wet periods, fresh water flows past the TR station through Little Madeira Bay and into northeast Florida Bay. During dry periods, salt water from Florida Bay can migrate into Taylor Slough, resulting in high-salinity levels at the TR station.

Salinity has been recorded at TR by Hydrolab[®] sondes at ten-minute intervals since July 14, 1988, with numerous periods of days to weeks of missing data, particularly at the beginning of the data record. As part of the modeling work, a salinity time series for the TR station for the period 1970 through 2002 was constructed (Marshall 2004a, 2005). The historical reconstruction is based on continuous salinity-monitoring data, which was available beginning in October 1988. In addition, the existing multivariate linear regression (MLR) salinity model was used to predict salinity for the period from 1970 to 1988 (Marshall et al. 2004). Data from these two sources were combined to create the historical reconstruction. This procedure is described in more detail in Chapter 5. The daily value salinity model is as follows:

TR salinity = 83.17 - 15.09CP[lag4] + 0.835Kwwatlev - 7.83(P33-P35)[lag1] - 4.34(P33-P35)[lag4]

where:

CP = stage (feet NGVD) at Craighead Pond

Kwwatlev = Key West water level (MSL)

P33 = stage (feet NGVD) at P33

P35 = stage (feet NGVD) at P35

Lag1 = one-day lag

Lag4 = four-day lag.

Ideally, the historical reconstruction should be applied on a monthly time scale (consistent with FATHOM). Thus the daily simulated values produced by the Taylor River MLR model were averaged to monthly values. Details on model development can be found in Marshall et al. (2004a, ECT, Inc 2005). Efficiency (a measure of the percentage of variance that is explained by the model variability) of the monthly Taylor River MLR salinity model is 84 percent (**Table 10**). The Taylor River model predictions compare reasonably well with observed values for the period 1988–2000, when observations exist (**Figure 31**).

Table 10. Summary of Uncertainty Statistics for the Monthly Taylor River MLR Salinity Model.

Station, monthly values	mean sq error (mse), psu	root mse (rmse), psu	adj R-sq	mean error, psu	mean abs error, psu	max abs error, psu	Nash-Sutcliffe Efficiency
Taylor River	12.71	3.56	0.84	-0.49	2.63	9.34	0.84
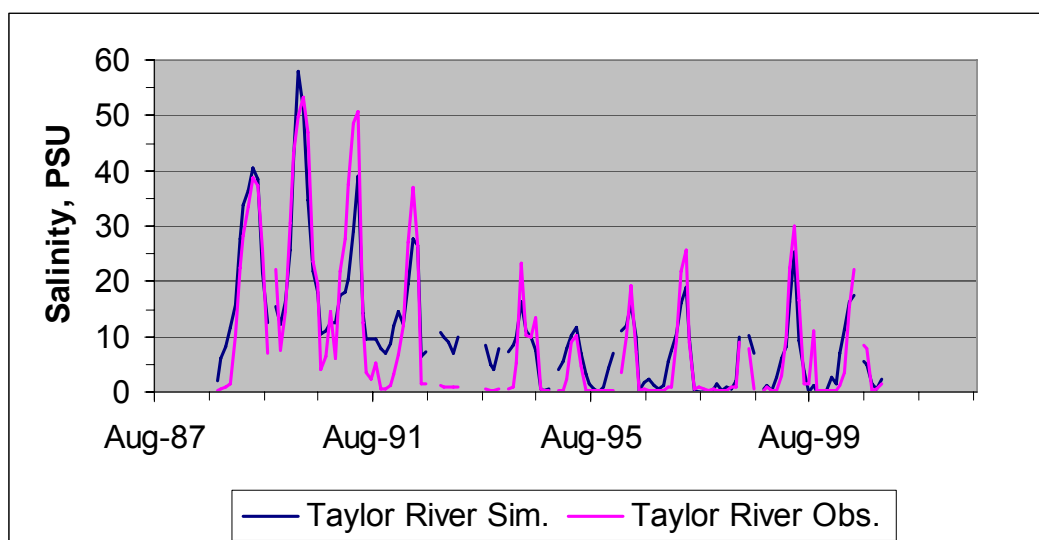
Figure 31. Predicted and Observed Average Monthly Salinity at the Taylor River (TR) Site. The highest error is associated with relatively short periods at onset of the wet season.

Examination of the daily and monthly plots and the daily uncertainty statistics indicates that the daily simulated values have an error of about 4.5 psu (Marshall 2004a). Some daily values may be as much as 10–15 psu in error during the month of May and, to a lesser extent, April, June, August and September because of interannual variability in the onset of the wet season. As with the daily values, monthly average salinity values are typically within 4 psu of observed values. Because of the potential for large residuals, particularly at the daily level, the following model limitations are apparent:

- The highest variability is associated with the relatively short period in which the dry season is ending and the wet season is beginning; the exact date and extension of this transition are not predictable (**Figure 31**).

- Measured flow in Taylor Slough can cease for relatively long periods, and so salinity simulations have the potential for high variability among extended low flow periods. Unfortunately, the reconstruction period contains two periods of extended low flow (namely, 1970–1974 and 1985–1990), and the 1970–1974 reconstruction should be viewed with this in mind. Observed values during the 1985–1990 dry period reached 60 psu, and maximum values during the 1970–1974 period probably also reached into that range.

ESTIMATING ECOLOGICAL EFFECTS OF WATER LEVELS AND FLOWS FOR MFL DEVELOPMENT

The ecologic characteristics of estuaries are strongly related to the influx of fresh water and associated materials from their watersheds (Day et al. 1989). Foremost among the influences of this watershed linkage is the effect of freshwater flow on the range and variability of salinity within estuaries. Salinity is a primary determinant of the species composition of communities and strongly influences functions of these communities (Sklar and Browder 1998). Altering the freshwater flow can also change the supply of nutrients to the estuary, thereby affecting estuarine productivity and habitat quality and availability.

Organisms living in the estuary have characteristic salinity tolerances and optimal salinities. Thus Florida Bay's salinity regime will determine how well these organisms can function, whether motile organisms will move out of the estuary to seek habitat offering more suitable conditions and whether certain other organisms will perish. Individual organism and population functions, in turn, determine the health of the entire ecosystem. If individual species are impaired by salinity stress, other components of the system that depend on them are endangered as well, resulting in a wider degree of systemic impairment of the ecosystem. For instance, a decline in the abundance or quality of seagrass habitat will have a detrimental impact on fauna that utilize this habitat. A decline in populations of small forage fish or invertebrates will have a detrimental impact on publicly recognizable sport fish populations. In this manner, the detrimental effects of salinity can cascade through the ecosystem.

All estuarine organisms are physiologically affected to some degree by the salinity level and by the rate of salinity change within an estuary. At extreme levels or with very rapid changes, salinity stress can be directly lethal to organisms, causing death in a relatively short time. Less extreme salinity stress may not be lethal but may nevertheless be just as important to the ecosystem; sublethal effects can include decreased growth and reproductive success, yielding a slow decrease in populations and changes in the structure and function of the food web.

Responses by animal species to changing freshwater inflow are not simply a matter of physiological tolerance. For instance, an important function of freshwater input is the seasonal appearance of a low-salinity signal that guides migrating organisms toward the nursery grounds in the wetlands (Shaw et al. 1985). Shrimp, certain fish species (such as menhaden and mullet) and other nekton have been shown to follow the salinity gradient toward a freshwater source, where they seek shelter to spawn or to complete their life stages and to consume special diet items while growing (Day et al. 1989). Without the appearance of the low-salinity signal at some distance from the freshwater source, offshore resident species may be disconnected from their inshore spawning and nursery grounds, resulting in reduced fisheries productivity or even in the demise of the species in that area.

The spatial expanse of estuarine conditions is also important in considering the potential ecologic effects of water levels and flows. The estuarine zone is a region of intermediate salinity created by the mixing of fresh and salt water and, absent freshwater inputs, the estuary would eventually change into a marine and hypersaline system. As the amount of freshwater input declines, the areas characterized by estuarine salinities generally diminish, resulting in less estuarine habitat and reduced area for feeding, fishing and spawning, processes that depend on the estuarine environment. Browder and Moore (1981) and Sklar and Browder (1998) emphasized the importance of the overlap of estuarine conditions and appropriate habitat (such as SAV or mangrove prop roots) for animal species. Decreases in the area of overlap, either by changes in habitat quantity or quality or by the occurrence of salinity conditions inhospitable to fauna, will decrease these faunal populations and ecosystem productivity. Furthermore, many animal and plant processes are not linear with respect to space; certain minimum areas and spatial

configurations (such as corridors) are required in order for some processes to occur (Micheli and Peterson 1999). Examples of spatial requirements are range area for mobile organisms, minimum predator-prey encounter areas, minimum refugia area for protective habitat and minimum sustainable seagrass patch size. Freshwater flows and salinity affect such biotic behavior and interactions both directly and indirectly by setting the spatial scale at which these processes occur. Thus, in addition to their direct salinity effects on biological organisms, changes in freshwater flow result both in systemwide changes in the physical size of the entire estuarine ecosystem and in local changes in spatial dimensions required for many ecologic processes.

The present section describes five types of analyses that were used to evaluate the ecologic effects of salinity conditions that will be used provide a basis for Florida Bay MFL criteria recommendations.

- General literature and data on important Florida Bay species were examined in order to determine the ecologic significance of these species and the environmental (salinity) conditions required for their survival.

- Field data from Florida Bay submerged aquatic vegetation (SAV) beds were analyzed and summarized to show, when possible, the statistical relationships among environmental conditions, distribution, cover and density.

- Analysis was performed on the results from mesocosm experiments on environmental tolerances and physiological responses of Florida Bay's SAV species.

- Modeling analysis of the field and mesocosm data was performed using a seagrass simulation model developed and calibrated specifically for Florida Bay. The model shows the predicted behavior of the seagrass community in response to different flow and salinity regimes.

- Statistical models were developed specifically for Florida Bay fish and invertebrate species to show the relationships among faunal densities, environmental parameters and seagrass composition and density.

The results of these five lines of analysis show that when the Everglades–Florida Bay transition zone and northeastern Florida Bay are exposed to marine and hypersaline conditions, biota are negatively affected and habitat is lost. A minimum freshwater inflow standard is critical for Florida Bay in order to ensure survival of critical ecosystem functions and species. The evaluation described in the following pages identifies 1) an individual species (*Ruppia maritima*) that is an overall indicator of the freshwater SAV community in the transition zone and 2) when monthly average salinities in the transition zone increase above 30 psu, the freshwater SAV community in this zone is lost and marine salinities may persist downstream for several months, resulting in adverse changes to seagrass communities in northeastern Florida Bay.

Everglades–Florida Bay Transition Zone and Its Submerged Aquatic Vegetation

The Everglades–Florida Bay transition zone is an ecotone containing numerous creeks, ponds, lakes and wetlands that include mangrove swamps and saline marshes (**Figure 23** and **Figure 32**). Hydrologic conditions in this zone are influenced by sheet flow and seepage of fresh water from the Everglades and by the intrusion of water from Florida Bay driven by wind and to a lesser degree by astronomical tides. Along the northeast and north-central Florida Bay coast, water exchange between the bay and the transition zone occurs in creeks that cut through a low-lying coastal ridge.

This transition zone is a major component of the greater Everglades ecosystem, with ecologic links to both Florida Bay and the freshwater Everglades, and it is a focus of the Comprehensive

Everglades Restoration Plan (Davis et al. 2005). In particular, the mangrove ecotone is an area that historically supported large populations of wading birds and waterfowl by providing a food base in the ponds and marshes and a place for rookeries in the nearby mangrove forests. These bird populations greatly decreased in the last century, probably in association with the hydrologic alteration of the Everglades and increased salinity in the transition zone (McIvor et al. 1994, Davis et al. 2005), but this zone still supports critical populations, including wood storks (an endangered species) and roseate spoonbills (a Florida species of special concern). This zone is also of special importance because it is the home of most remaining American crocodiles (an endangered species) in the United States (Mazzotti 1999).
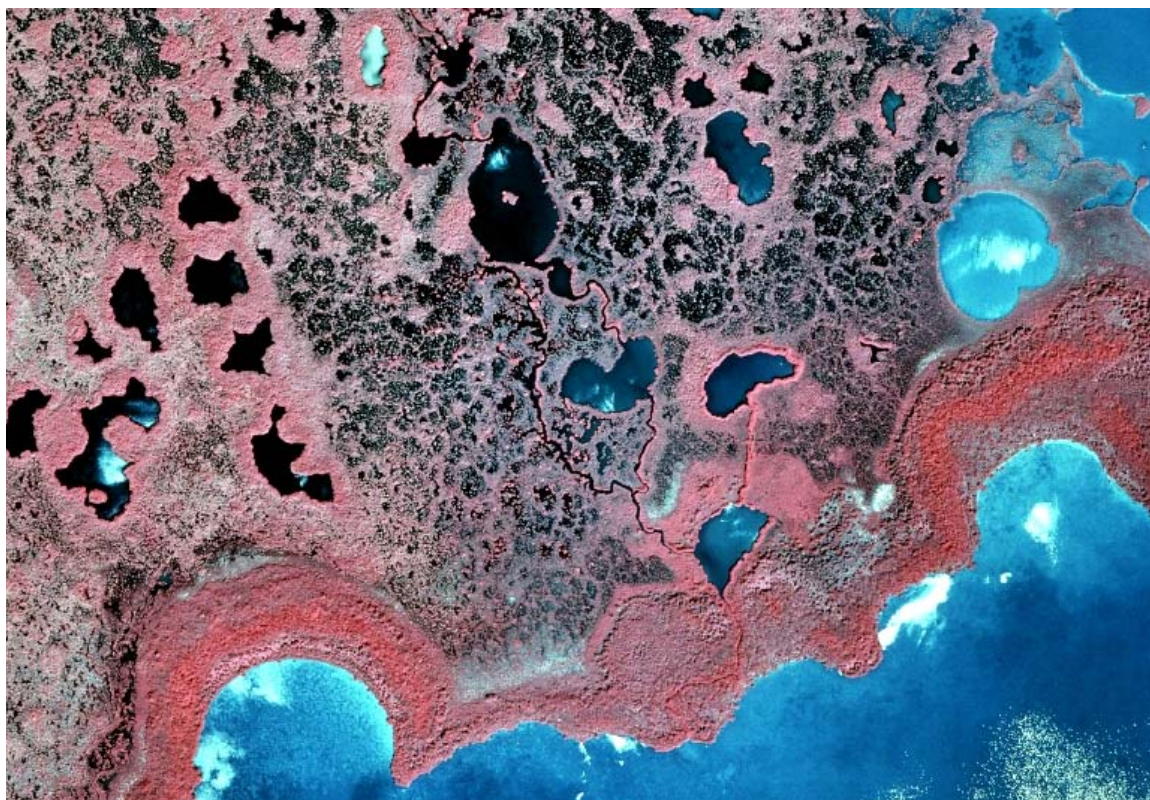
Figure 32. Satellite Image of the Salinity Transition Zone near Taylor River. The area shown by this image is north of Little Madeira Bay (see **Figure 23**); color patterns show the heterogeneity of the landscape, including many ponds; dark red-pink areas fringing the shorelines of the bay and ponds are canopies of red mangroves; the distance from the southern [lower] shoreline to the northern [upper] edge is about 2 km).

Geomorphology and the salinity gradient within the transition zone from the Everglades to the bay are primary factors structuring ecologic zones within the transition zone, as described by Ross et al. (2000, 2002). Wetlands found at the boundary of the bay and bordering numerous mangrove creeks and ponds within about five kilometers of the bay are dominated by *Rhizophora mangle* (red mangrove) trees (**Figure 32**). Adjacent interior saline wetlands are also dominated by red mangroves, but these mangroves are dwarfed and in the form of shrubs because of nutrient limitation (Koch and Snedaker 1997). Toward the interior of the transition zone, marshes contain a mixture of mangrove shrubs and graminoid vegetation (grasses and grasslike plants, such as sedges: mostly *Eleocharis* spp. and *Cladium jamaicense*). Much of this zone has low productivity and sparse vegetation and appears as an area of high reflectance in satellite images (a "white zone," in the term of Ross et al. 2000). The area of this white zone has increased during the past fifty years, with the interior boundary extending inland by up to four kilometers—a shift

hypothesized to be the result of increased saltwater intrusion associated with reduction of freshwater input from upstream and occasioned by changes in water management. *Cladium jamaicense* (sawgrass) dominates the freshwater boundary of the transition zone.

Transition zone wetlands are an important foraging area for wading birds. Lorenz (1999) has described productivity patterns—vis-à-vis salinity and water levels—of fish that constitute the forage base for such birds as the roseate spoonbill, finding that the density and biomass of this forage assemblage, which is dominated by members of Cyprinodontidae (killifishes), Poecillidae (livebearers), Gobiidae (gobies) and Atherinidae (silversides) tend to decrease with increasing salinity and increase with longer, more-stable hydroperiods. Experiments by Rowe and Dunson (1995) suggest that these results could stem from an interaction of fish concentration (prey density) and salinity factors: in their work, growth and survival of *Cyprinodon variegatus* (sheepshead minnow), a common species in the mangrove transition area) were reduced at higher salinities (32 psu) when combined with high fish density. Foraging success of wading birds is highly dependent upon the decreasing water levels in the early dry season, which concentrate prey for the birds (Frederick and Spalding 1994, Davis et al. 2005).

Submerged aquatic vegetation in transition zone waters is important as a base of the food web and as habitat. In particular, *Ruppia maritima* (widgeon grass), the dominant vascular plant of this SAV community, is known to be an important food source for wintering waterfowl, including coot, scaup, widgeon and pintail (Kushlan et al. 1982). The abundance of these waterfowl populations in transition zone ponds has greatly decreased in the past fifty years (Davis et al. 2005), a drop hypothesized to be the result of declines in SAV productivity and cover because of increased salinity and prolonged periods of high-salinity conditions within naturally oligohaline and mesohaline ponds (Morrison and Bean 1997, Montague et al. 1998, Davis et al. 2005). Isotopic studies of the transition zone food web (in Whitewater Bay) suggested that *Ruppia* was a major food source for forage fish and invertebrates that were the food base for gray snapper (Harrigan et al. 1989).

Field observations suggest that *Ruppia maritima* may provide important habitat for small forage fishes inhabiting lower-salinity areas (Garcia and Vierira 1997, Duffy and Baltz 1998, Castillo-Rivera et al. (2002, 2005). Moreover, Rutherford et al. (1986) demonstrated higher densities of juvenile snook in areas of western ENP estuaries dominated by *Chara* sp., *Ruppia maritima*, and other low-salinity vegetation. Ley (1992) found high densities of fishes and foraging water birds in transition zone of the southeast Everglades, with dense *Ruppia* and associated macroalgae. As this SAV disappeared during the 1989-1990 drought period (when salinity rose well over 50 psu in the mangrove transition zone ponds and creeks), the fish community became depleted and resulted in fewer water birds foraging in these areas. These results support that *Ruppia* and other transition zone SAV provide an important habitat function for the fish and avian community of northeastern Florida Bay.

The food web of transition zone ponds and creeks also supports the endangered American crocodile (Kushlan and Mazzotti 1989, Mazzotti 1999), which in the United States is limited in distribution to the southern tip of Florida and the upper Florida Keys. Crocodile habitat once extended from central Florida southward, but now more than two-thirds of the nests of this federally listed endangered species are found along the Florida Bay coast (USFWS 1998). The area comprising the northeast coast of Florida Bay and transition zone has been designated by ENP as a crocodile refuge, in order to protect these nests. Crocodile nesting success or failure is related to factors such as flooding, desiccation, salinity and predation (Mazzotti 1989, USFWS 1998). The critical time for hatchlings is from late summer through fall, a period in which the historic system delivered greater volumes of fresh water into areas of crocodile habitat (McIvor et al. 1994). Hatchling crocodiles have higher relative metabolic demands and less ability to osmoregulate than do their adult counterparts. Seeking fresh water can be energetically expensive for young crocodiles. Salinity greater than 20 psu in nearshore nesting areas is considered detrimental to the growth and survivorship of young-of-year crocodiles (Mazzotti et al.

1986, Moler 1991, USFWS 1998). In addition, spatial and temporal extension of low-salinity conditions could increase forage fish density and biomass Lorenz (1999). Increasing the forage base (such as transition zone fishes) for pre-adult crocodiles may increase crocodile growth and survivorship in nursery areas around Florida Bay (Mazzotti, personal communication).

Background and Evaluation of the Literature

The transition zone vegetation complex of Florida Bay is important to fauna as a food source and as refuge, supporting a number of faunal species that inhabit the zone either transiently or as resident species (Ley and McIvor 2002, Lorenz et al. 2002). These plants also perform important ecosystem functions outside of the transition zone, supplying detritus for export to the greater estuary, thereby supporting the provision of food for other nekton (Zieman 1982, Snedaker 1989). Primary production in the brackish transition zone also provides a source of dissolved organic compounds distributed within the zone and into Florida Bay, potentially supporting a microbially based food web (Snedaker 1989, Lavrentyev et al. 1998). Furthermore, SAV can sequester nutrients and enhance nutrient retention within the transition zone, which may be important for good water quality in the larger Florida Bay (Rudnick et al. 1999, Davis et al. 2005).

Only a small group of vascular plant and macroalgae species are adapted to grow in the wide-ranging and rapidly changing salinity conditions of the transition zone. Most freshwater plants cannot survive salinity exposure, particularly above mesohaline levels. Likewise, most true seagrasses cannot survive the sustained (several months) freshwater conditions common in the transition zone. If salinity exceeds the tolerance of the few species adapted to this zone, loss of SAV can occur, leaving the benthic habitat as bare, unvegetated substrate (Morrison and Bean 1997, Ley, 1992, Montague et al. 1998).

Submerged vegetation in the ponds and channels of the Florida Bay transition zone has been studied relatively little, usually in localized areas (Montague et al. 1998, Morrison and Bean 1997, Tabb and Manning 1961, Tabb et al. 1962, Zieman 1982). Likewise, the mapping of species composition and distribution across the northern coast of Florida Bay has not been performed comprehensively. Available data show that the vegetation of the transition zone is dominated by characteristic plants common in fresh water and brackish water, including *Chara* spp. (muskgrass, a multicelled macroalga), *Utricularia* spp. (bladderwort), *Ruppia maritima* (widgeon grass, a bushy, fanlike underwater freshwater plant that has a high tolerance for salinity and alkalinity) and *Halodule wrightii* (shoal grass, a seagrass that can withstand a wide range of temperatures and salinities).

A Florida Bay SAV background study for the present report (Battelle 2004) provides information on each of the major vascular SAV species found in the Florida Bay transition zone, including *Ruppia maritima* (widgeon grass) and *Halodule wrightii* (shoal grass) (for additional information on shoal grass, see also Doering et al. 2002).

The vegetation complex of the eastern bay's salinity transition zone has been monitored by National Audubon Society scientists since 1996, and the coastal bays of northern Florida Bay (such as Joe Bay and Little Madeira Bay) have been monitored since about 1995 by Miami-DERM, Madden et al. (2003) and the Fish Habitat Assessment Program (FHAP) of the Florida Fish and Wildlife Conservation Commission.

Tabb and Manning (1961) and Tabb et al. (1962) described the biota of northwestern Florida Bay and the Whitewater Bay and Coot Bay region in the 1950s. The salinity transition zone, represented by Coot Bay and eastern Whitewater Bay, was dominated by *Ruppia maritima* (widgeon grass) and *Halodule wrightii* (shoal grass), while northern Florida Bay was dominated by *Thalassia testudinum* (turtle grass) and Halodule. In low-salinity ponds and lakes of the western transition zone, Tabb and Manning (1961) found *Chara* (musk grass) to be predominant

where salinity was below 15 psu, while in the more variable salinities of Coot Bay, a distinct zonation occurred, as follows: 1) *Chara* at salinities below 12 psu, 2) *Ruppia* at salinities between 12 and 28 psu and 3) *Halodule* replacing both in areas of salinities greater than 28 psu. The same studies also described the seasonal succession of these plants, indicating that in Coot Bay, *Ruppia* dominated when salinity was below 15 psu but was replaced by *Halodule* during the dry season when salinity rose above 20 psu. During the wet season, both *Ruppia* and *Halodule* were reported to be replaced by *Chara* when salinity fell below 10 psu.

Morrison and Bean (1997) found a similar pattern in ponds of the transition zone of the north-central Florida Bay: *Chara* was found to have an apparent maximum salinity tolerance in the range of 15 to 20 psu. Meanwhile, Montague et al. (1998) studied the transition zone of northeastern Florida Bay and found *Chara* in ponds and streams with a mean salinity of 6 psu. and *Najas* (waternymph) and *Utricularia* (bladderwort) at sites with a mean salinity of 2 psu, observing that both *Najas* and *Utricularia* increased in abundance as the wet season progressed.

Zieman (1982) noted that by the 1980s Florida Bay as a whole was undergoing a shift toward development of monospecific stands of *Thalassia*, with a general loss of *Halodule* and macroalgal species. He attributed this shift to reduction in freshwater inflow and elevation of salinity that had occurred in the previous two decades. Consistent with Tabb and Manning (1961), Zieman (1982) and Zieman et al. (1989) found that *Ruppia* grew well only in areas adjacent to freshwater inflow (Zieman 1982, Zieman et al. 1989). *Ruppia* was generally associated with stands of red mangrove and was located around the fringes of the ecotone in the eastern part of the bay. The vegetation complex in other areas of the northern bay and transition zone of the 1970s and 1980s was described as containing dense stands of *Thalassia* in the coastal bays and equally dense and monotypic stands of Halodule of up to 90 g dw m^{-2} in bays and ponds of the transition zone. Zieman et al. (1989) hypothesized that in the high-light, high-salinity environment created by reduced freshwater input, *Thalassia* thrived at the expense of *Ruppia* and freshwater macroalgae.

The distribution of SAV in Florida Bay has changed during recent decades, probably in response to both natural and human factors. Since the early 1990s, freshwater flow to the bay has increased, and the somewhat fresher salinity regime in the transition zone has most likely promoted an expansion of the freshwater and brackish-water plant assemblage, with reductions in *Thalassia* coverage in the immediate area of the transition zone (Miami-DERM 2005). Montague et al. (1989) found *Ruppia* to dominate twelve northeastern Florida Bay sites sampled in 1986 that experienced highly variable salinity fluctuations between 0 and 30 psu. *Ruppia* continues to dominate the Florida Bay transition zone as the primary rooted vascular plant, and several freshwater macroalgal species are also abundant in the region, notably *Chara* sp., *Najas* sp. and *Utricularia* sp. (Montague et al. 1989, Montague and Ley 1993, Morrison and Bean 1997, Miami-DERM 2005). The dominant macroalgal species in the transition zone are generally obligately oligohaline or prefer lower salinities and, despite *Ruppia*'s ability to tolerate high salinities, it appears to be outcompeted by true marine seagrasses at even intermediate salinities. In Florida Bay, *Ruppia* and the macroalgal complex are not found in areas seaward of the transition zone.

Overview of *Ruppia maritima*

Ruppia maritima, commonly known as widgeon grass, is distributed worldwide, occurring in temperate and subtropical estuaries, bays and lagoons and in inland saline lakes and wetlands. This angiosperm is recognized worldwide as an important food of migrant and wintering shorebirds, wading birds and waterfowl and is heavily used by fish in coastal wetlands (Kantrud 1991). Propagation and management of *Ruppia* have occurred for nearly 60 years in the southern and eastern United States, and comprehensive studies and literature reviews are available (Kantrud 1991 and references contained therein, Tyler-Walter 2001 and references contained therein). *Ruppia* has a well-defined ecologic niche. It grows poorly in water with low water clarity

or anaerobic sediments, but has specialized features enabling survival under a wide range of salinities and high temperatures beyond those tolerated by most other submerged angiosperms (Kantrud 1991).

Ruppia maritima serves many ecologic functions for a variety of organisms. The leaves and stems of *Ruppia* provide substratum and refuge for several species, and the rhizome and root system stabilize the sediment, transport oxygen from the leaves and oxygenate the sediment in the vicinity of the roots, changing the soil redox potential, sediment chemistry and oxygen levels (Verhoeven and van Vierssen 1978). The decomposition of *Ruppia maritima* leaves and stems supports a detrital food chain within this habitat, especially in temperate regions during autumn and winter. Suspension feeders and bottom feeders such as bryozoans, polychaetes, amphipods, bivalves and chironomid larvae may utilize the detritus produced from the decomposition of *Ruppia* leaves (Tyler-Walters 2001). Verhoeven and van Vierssen (1978) and Verhoeven (1980b) suggested that isopods and amphipods may feed directly on this plant. But *Ruppia*'s most important role in the food chain is the breaking down of decomposed leaves into fine particles of detritus suitable for suspension and incorporation into the detrital food chain (Verhoeven and van Vierssen 1978, Zieman et al. 1984, Harrigan et al. 1989, Kantrud 1991). The leaves of *Ruppia* are commonly colonized by diatoms and other epiphytes and commonly combine with floating mats of filamentous green algae (such as *Cladophora*) and *Chara*. The epiphytes and algal mats of *Ruppia* may be grazed by gastropods, amphipods, isopods and mysids (Tyler-Walters 2001). Faunal epibionts such as bryozoans and hydroids colonize *Ruppia* leaves and also may provide temporary substratum for juvenile anemones and bivalves and the larvae and pupae of aquatic insects (Verhoeven and van Vierssen 1978, Verhoeven 1980a, Boström and Bonsdorf 2000). Other organisms use *Ruppia* beds as habitat and shelter from predation. Small invertebrates are preyed on by mysids, shrimp and forage fish that utilize *Ruppia* habitat (Tyler-Walters 2001). Epifauna, small shallow infauna and larger infauna are probably the most common foods for fish (Harrigan et al. 1989, Montague et al. 1989). Specific studies regarding the epifaunal assemblages within the northeastern Everglades–Florida Bay transition zone are very limited, but Montague et al. (1989) conducted studies that indicated the importance of epifauna associated with the leaves and stems of submerged vegetation within this area. These researchers found a strong correlation of vegetation and benthic infauna in the ponds of the northeastern transitional zone. They could not definitively state, however, whether this correlation was related to the enhanced presence of food and cover that the SAV provide or related independently to the salinity variation causing low densities of both SAV and benthic infauna.

Distribution of *Ruppia maritima* in Relation to Salinity

Studies of the natural history of *Ruppia maritima* (widgeon grass) show the species to be not a true seagrass but a freshwater species unusually tolerant of salinity (McMillan 1974, Verhoeven 1975). Among estuaries worldwide, the species is found in salinities ranging from zero to full-strength seawater, although it is generally distributed and grows most rapidly where salinity is below 25 psu (Phillips 1960). The species commonly dominates in the brackish region of estuaries (Kantrud 1991) and appears to disappear from environments that change from low salinity to marine conditions (Murphy et al. 2003). The species is also common in saline inland lakes (for instance, in prairie potholes of interior North America), at much higher salinities (>100 g/L) in these environments than is typical in estuarine and marine environments (Kantrud 1991).

Ruppia populations have been studied in Texas, North Carolina and several locations in Florida, including Apalachee Bay, the Econfina River, Tampa Bay and Florida Bay (Battelle 2004). Populations are generally found in mesohaline areas, generally from 10 to 30 psu, although many of these areas experience variable salinities and it is difficult to know the salinity range encountered by plants in the field. Phillips (1960) found that *Ruppia maritima* occurred below 25 ppt in Tampa Bay. Iverson and Bittaker (1986) surveyed stations in river mouths of the Florida west coast over six years and found *Ruppia* to be prevalent in low-salinity areas. Koch and Dawes (1991) harvested plants in western Florida at salinities ranging from 2 to 14 psu and in

North Carolina at salinities between 6 and 30 psu. These reported salinity tolerances for *Ruppia* are similar to those found in Florida Bay and Whitewater Bay.

Ruppia maritima Population Dynamics in Relation to Salinity

The aforementioned *Ruppia* distribution patterns reflect the net effects of salinity and other factors on the growth, reproduction and mortality of *Ruppia* populations. An important distinction in plant ecology is 1) a population's physiological tolerance to salinity and 2) the population's actual distribution over a range of salinities in nature. The difference between the two may be attributable to reproductive failure, predation, disease, nutrient resource limitation or other similar factors and stresses. Studies in laboratories have attempted to ascertain exact salinity tolerances of *Ruppia* in mesocosms, and most have found that plants can tolerate very high salinities for limited periods. Lazar and Dawes (1991) found that plants from a Tampa Bay population survived well when exposed to 35 psu in mesocosms, and Koch and Durako (2004) found tolerance of adult plants up to 70 psu in mesocosm studies over a four-month period. In laboratory incubations, Murphy et al. [2003] found a significant depression in *Ruppia* photosynthesis at 40 psu, but these were short-term experiments and the plants may not have acclimated to experimental conditions.

The optimal salinity for *Ruppia* growth appears to be < 20 psu (Kantrud 1991), but studies of populations in Florida Bay and in other areas indicate that established *Ruppia* plants can tolerate higher salinities and even hypersalinity for extended periods. A factor that may increase *Ruppia* mortality is not simply the magnitude of salinity but the rate of change of salinity. In creeks and small ponds of the Florida Bay transition zone, where salinity can drop very rapidly, Montague and Ley (1993) found that SAV biomass was more closely correlated with salinity variance than with salinity magnitude. Rapid fluctuations have also been reported to kill *Ruppia* when salinities rose > 18 g/L in a few weeks (Verhoeven 1979).

The collective evidence on the relationship between *Ruppia* reproduction and salinity points toward the impact of the magnitude, timing and duration of high-salinity events (marine and hypersaline) on Ruppia populations, in part perhaps because the seeds appear to be sensitive to salinity levels. Dunton (1990) found that *Ruppia* populations in two different Texas lagoons ranging in salinity from 0 to 25 psu and 32 to 38 psu were equally productive; however, the population at the high-salinity site in the Nueces River had an overwintering form, while the low-salinity site population in the Guadalupe Estuary did not. These observations suggested that *Ruppia* seeds may be sensitive to high salinity, requiring the plants to propagate vegetatively at the high-salinity site.

Ruppia seeds generally overwinter for one season before germinating the following spring (Phillips 1960); therefore, the spring-summer period of germination and seedling development may represent a period during which the appropriate salinity regime is especially important. It is not known if periods of high salinity kill seeds (likely not), nor is the length of time seeds remain viable in sediments well known (probably between one year [Hanlon and Voss 1975] and three years [Kantrud 1991]). But perhaps the most important reason that *Ruppia* populations are uncommon in estuarine areas with salinities frequently above 30 psu is reproductive failure. Flowering, and hence seed production, is reported to occur only at salinities below 30 psu (Kantrud 1991), a finding consistent with the observations made on Florida's west coast by Iverson and Bittaker (1986). Germination of *Ruppia maritima* seeds has been reported to be greatly reduced where surface sediments contain more than 20 g/L soluble salts or where sodium chloride concentrations in the water exceed 15 g/L (Kantrud 1991).

Interactive Effects of Salinity and Other Factors on *Ruppia maritima*

Site-specific differences may also be important for *Ruppia maritima* germination. In laboratory work, Koch and Dawes (1991) found germination differences in seeds obtained from estuaries in North Carolina versus seeds obtained from estuaries in Florida. Seeds obtained from Pamlico Sound, North Carolina, had an earlier germination time (25 days) in fresh water and a significantly higher germination rate at all salinities tested (0, 15, 30 psu) as compared with the seeds from the Weeki Wachee River, Florida (germination time = 35 days in fresh water). Germination of the Florida seeds was time delayed even further (> 30 to 68 days) and < 3 percent successful at 15 psu and did not occur at all at 30 psu (80 seeds used per treatment).

Water temperature and seasonal temperature fluctuations also influence reproductive success (particularly germination) and may be important considerations in *Ruppia maritima* reestablishment, particularly in environments with widely fluctuating or high salinity. Harrison (1982) found that seedling success differs from year to year, especially in response to variations in conditions (weather) in early spring, when germination and establishment occur. Seeliger et al. (1984) hypothesize that optimal germination conditions for *Ruppia* vary from latitude to latitude because of temperature differences. Kantrud (1991) provides a comprehensive discussion of germination and growth potential of Ruppia in mild climates and suggests that temperature plays a role in the life strategy of the species, noting that germination rates of seeds are higher for those kept at lower temperatures in waters where salinity ranges up to 26 g/L than for those kept at higher temperatures in fresher (< 3.5 g/l) waters.

In work comparing seeds from a Florida estuary (mild seasonal water temperature fluctuations) with seeds from a North Carolina estuary (higher seasonal fluctuations), Koch and Dawes (1991) found differences in germination over a range of temperatures tested (salinity of 0 psu). The North Carolina seeds exhibited a significantly higher rate and number of germinations than did the Florida seeds at all temperatures tested (17, 23 and 29° C). The authors suggest that the smaller seasonal fluctuation in temperature in Florida, with favorable temperatures for growth throughout the year, may account for the slower germination rate of seeds from that area. In contrast, the North Carolina seeds may be adapted to take advantage of a much shorter growing season. Further illustrating the importance of considering temperature in milder climates in combination with higher salinity fluctuation, Seeliger et al. (1984), in work using seeds obtained from Patos Lagoon estuary in southern Brazil, found that the best germination response (> 10 percent) was obtained at lower salinities (< 20 psu) and after 12 months of cold storage (7° C). Water temperatures do not normally get very low in the Everglades–Florida Bay transition zone, but the North Carolina and the Brazil studies both indicate that exposure to the combination of lower water temperatures and low salinity may be important for successful germination. Given the climate in southern Florida, fairly low germination rates throughout the year for *Ruppia maritima* would be expected in a brackish environment such as the transition zone, and reestablishment from seeds after a significant stress (such as a hypersaline period) may be precluded or delayed because of the combination of salinity and relatively high temperature.

Some field evidence also indicates that reestablishment of *Ruppia* in the Everglades–Florida Bay transition zone may be negatively affected by high temperature and high salinity. In the course of a two-year field study, Montague et al. (1989) observed that relatively dense *Ruppia*-dominated vegetation disappeared in a pond along Snook Creek (a small tributary to Joe Bay, which is a northeastern embayment of Florida Bay). The SAV had been abundant in March through May of 1986 but disappeared thereafter for the remainder of the study period (through September 1987). Salinity rose at this site in March through May from 13 ppt to 26 ppt and by June had dropped to 1 ppt, where it remained until August of that same year. Dense mats of the filamentous blue-green alga *Lyngbya* appeared in Snook Creek following the disappearance of the macrophytes (including *Ruppia*); SAV was no longer observed for the duration of the study period. The researchers cited either high salinity, salinity shock (a sudden salinity drop) or a combination of the two as the most likely cause. This observation illustrates potential difficulties in the

Everglades–Florida Bay transition zone with regard to reestablishment of vegetation after that vegetation's demise. Apparently, for an entire year thereafter, two months of low salinity in the summer months was not sufficient time or condition for any SAV, including *Ruppia maritima*, to become reestablished in this pond area. This finding suggests that once SAV has disappeared, a sufficient duration of a low salinity conditions with appropriate seasonal timing (temperature) must be maintained the Everglades–Florida Bay transition zone in order to promote reestablishment.

Another interactive factor that potentially influences reproductive success is dissolved-oxygen availability, which varies as a function of temperature and salinity. Kantrud (1991) reported that oxygen scarcity, as indicated by a redox potential of -300 mV, retards germination. Senescence and loss of stems seems to coincide with increases in hydrogen sulfide in the sediment and may be a factor that helps explain decreased germination in hot summer months when sulfate reduction rates (and hence the production of hydrogen sulfide) are likely to peak. The saline ponds of the transition zone typically contain sediments with high concentrations of organic matter (> 10 percent of dry weight) and low concentrations of iron, resulting in high sulfide concentrations (Koch et al. 2001).

Laboratory Analysis

Salinity's effects on *Ruppia maritima* were examined in Florida Bay and the Florida Bay transition zone using a combination of controlled laboratory studies and field data. Mesocosm experiments were conducted to test the response of *Ruppia* to salinity and temperature (Koch and Durako 2004); effects on adult plant growth and survivorship, as well as on seed germination and seedling development, were measured. Results showed that in the mesocosm, adult *Ruppia* plants tolerated salinity as high as 70 psu for up to four months. However, this result was inconsistent with maximum salinities at locations where *Ruppia* is observed in the transition zone, suggesting that in the field, *Ruppia* distribution is not a simple function of adult plant salinity tolerance. Additional important controls on distribution may include nutrient limitation, thermal stress, light limitations, substrate incompatibility, disease and grazing, at higher salinities. These factors may act as controls independently or in concert with salinity.

One line of evidence that may explain the confinement of *Ruppia* beds to areas of lower salinity (less than 25–30 psu) is shown in studies of seed germination at different salinities; a laboratory germination study using *Ruppia maritima* seeds from Florida Bay (Koch and Durako 2004) showed that seed germination is inhibited at salinities above intermediate levels (**Figure 33**). Approximately one-third of the seeds that were incubated at low to intermediate salinities successfully germinated, but germination did not occur at any salinity treatment higher than 30 psu, even when salinity was slowly increased to allow time for acclimation. Without slow acclimation, germination did not occur at salinities higher than 20 psu. This outcome is consistent with results from an earlier germination experiment by Koch and Dawes (1991), who found lower rates of germination at 15 psu than in fresh water and no seed germination at all at 30 psu. Sustaining a plant population requires not only that adults can physiologically tolerate the environment but also that they can successfully reproduce in that environment.
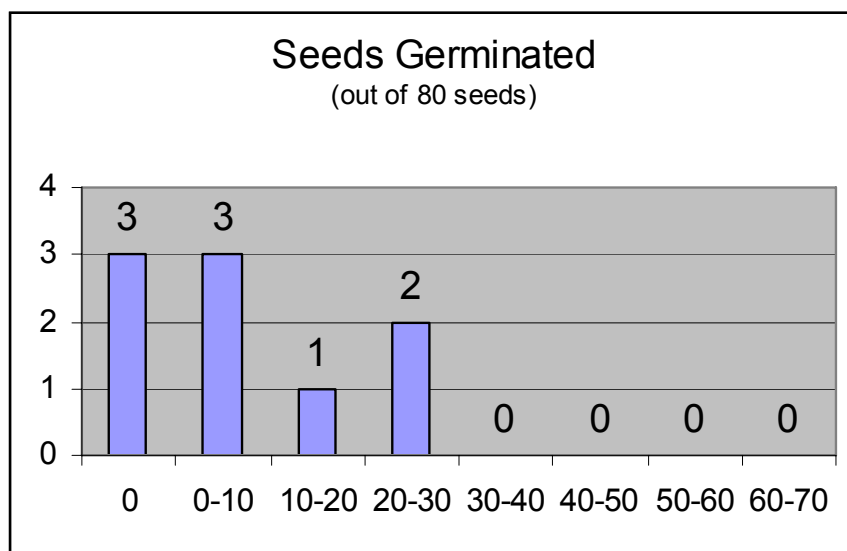
Seeds Germinated
(out of 80 seeds)

Salinity class	Seeds germinated
0	3
0-10	3
10-20	1
20-30	2
30-40	0
40-50	0
50-60	0
60-70	0

Figure 33. Salinity's Effect on *Ruppia maritima* Seed Germination. Number of seeds germinated (out of 10 seeds per treatment) during four-month incubations. Eighty seeds were incubated in eight salinity classes in the experiment (Koch and Durako 2004).

Analysis of Field Data

Field data from the 1996–2004 period (National Audubon Society, Frezza and Lorenz, unpublished) show that percentage cover by *Ruppia maritima* decreases with increasing salinity in Florida Bay. Ruppia as was dominant cover, but declined significantly ($p < 0.001$ from analysis of variance [ANOVA]) in waters above 25 psu. Data were collected along three transects through the transition zone and into the bay: 1) Taylor River into Little Madeira Bay, 2) from a creek that flows into northeastern of Joe Bay, through Joe Bay, and into Trout Cove and 3) along Highway Creek flowing into Long Sound (**Figure 34** and **Figure 35**). The first transect (Taylor River to Little Madeira Bay) corresponds to the representative transition zone gradient described in this report. Salinity varies more rapidly than plants can respond in terms of cover, and so *Ruppia* cover was also compared with the average salinity during the 30 days prior to the day of the sample, with virtually the same results.
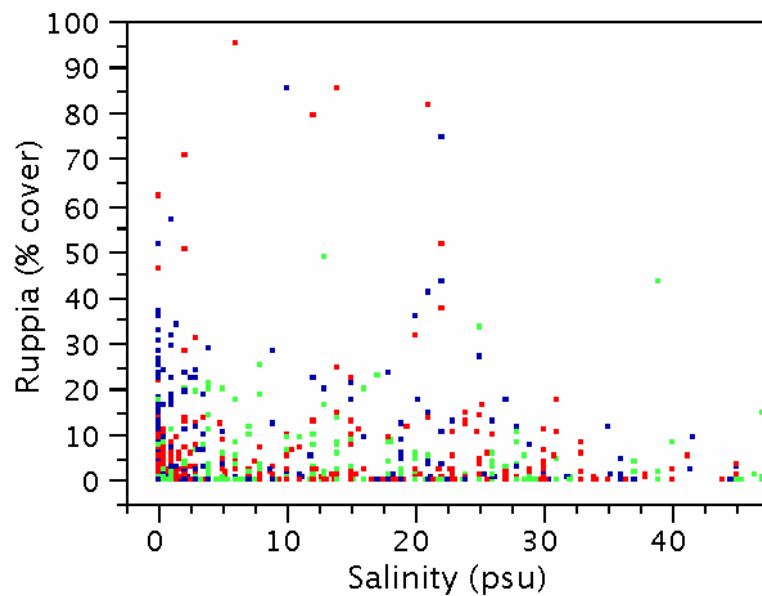
Figure 34. *Ruppia maritima* Cover in Relation to Corresponding Instantaneous Salinity in the Everglades–Florida Bay Transition Zone. blue = Taylor River transect (corresponds to the representative transition zone gradient described in this report – see **Figure 23**), red = Joe Bay transect, green = Highway Creek transect) Data from National Audubon Society, Frezza and Lorenz (unpublished).
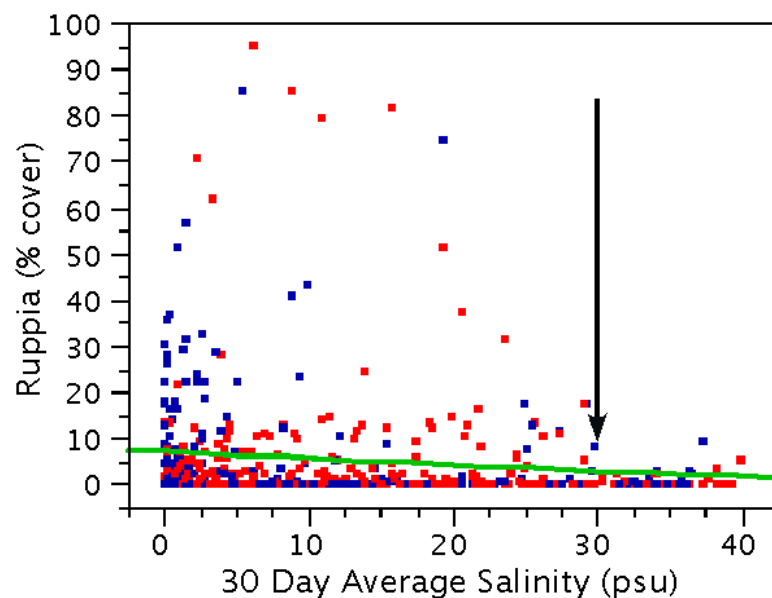

Figure 35. *Ruppia maritima* Cover in Relation to Corresponding 30-Day Average Salinity in the Transition Zone. Data shown were extrapolated from continuous monitoring data for the Florida Bay–Everglades transition zone; arrow points to 30 psu level, where *Ruppia maritima* cover decreases to below 5 percent; green line is regressed trend line that indicates decreasing *Ruppia maritima* cover with increasing salinity (p=0.0084, 1.d.f); blue = Taylor River transect; red = Joe Bay transect. Data from National Audubon Society, Frezza and Lorenz (unpublished].

To estimate average 30-day salinities, instantaneous salinity measurements at *Ruppia* sampling sites were extrapolated using regressions of salinity measured at the SAV sites against daily salinity measurements recorded at the nearest SFWMD, ENP and USGS continuous-monitoring platforms (**0** A trend line through the relationship between average salinity and *Ruppia maritima* cover suggests that average cover decreases with salinity (p=0.008, 1 df) (**0** The 30-day salinity data were also grouped into several discrete categories— 0–10 psu, 10–20 psu, 20–30 psu and >30 psu—and were compared with corresponding SAV cover. The 0–10 psu category exhibited the highest percentage of cover, and the mean cover decreased with each increasing salinity range. ANOVA showed that the cover means were statistically different (p=0.0272, with 3 df) for each category, and a pair-wise t-test identified that the >30 psu category was significantly lower than the 0–10 psu category (p < 0.05). In summary, *Ruppia maritima* cover appears to be significantly reduced at salinities above an average of 30 psu.

Table 11. Regression Equations Used to Calculate 30-Day Average Salinity Values in the Transition Zone . Equations were developed for each of the National Audubon Society sites along the Taylor River and Joe Bay transects; daily salinity data used for the extrapolations were obtained were from the ENP Argyle Henry (ENP-AH), USGS Taylor River ([USGS-TM), and the SFWMD Joe Bay (SFWMD-JB) platforms; for each site, instantaneous salinity data were regressed against the daily salinity measurement from the associated platform to create the extrapolation equations.

Station	Platform (Agency-Site)	Extrapolation Equation	R^2
TR1	ENP-AH	TR1 = -0.446+0.972*AH	0.967
TR2	ENP-AH	TR2 = -0.342+0.984*AH	0.976
TR3	ENP-AH	TR3 = -4.201+6.240*Sqrt(AH)	0.964
TR4	ENP-AH	TR4 = -3.573+6.429*Sqrt(AH)	0.955
TR5	ENP-AH	TR5 = -2.186+6.284*Sqrt(AH)	0.938
TR6	USGS-TM	TR6 = 5.922+0.832*Sqrt(TM)	0.912
JB1	SFWMD-JB	JB1 = -0.556+0.028*JB^2	0.848
JB2	SFWMD-JB	JB2 = 0.695+0.027*JB^2	0.850
JB3	SFWMD-JB	JB3 = -1.221+0.929*JB	0.910
JB4	SFWMD-JB	JB4 = 0.304+1.001*JB	0.907
JB5	SFWMD-JB	JB5 = 3.479+0.964*JB	0.856
JB6	SFWMD-JB	JB6 = 6.475+0.851*JB	0.806

Similar patterns of SAV cover and salinity were observed for the macroalga *Chara* (muskgrass), with an apparent salinity threshold near 30 psu (**Figure 36**). Patterns for two other common macroalgae, *Najas* (waternymph) and *Utricularia* (bladderwort), indicated a lower salinity tolerance; little cover was found above 15 psu (**Figure 37** and **Figure 38**).
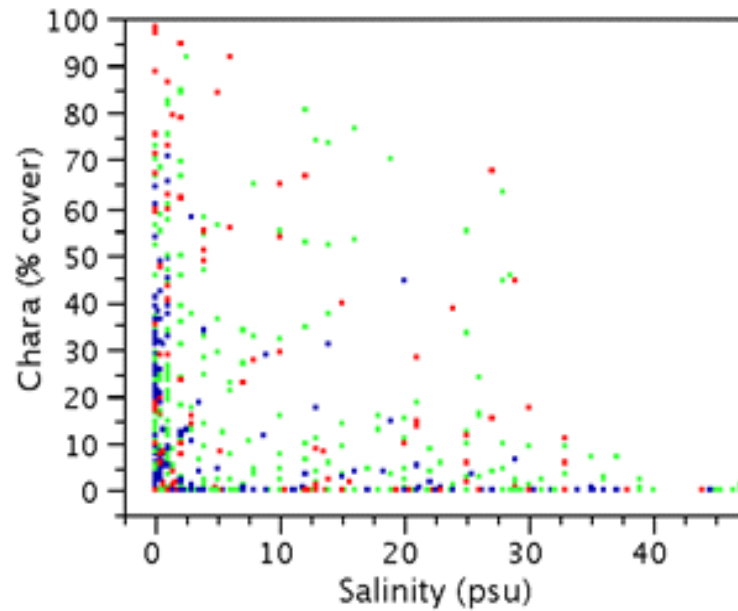
Figure 36. *Chara* (Muskgrass) Cover in Relation to Instantaneous Salinity in the Transition Zone. Blue = Taylor River transect (corresponds to the representative transition zone gradient described in this report [see Figure 23]); red = Joe Bay transect; green = Highway Creek transect. Data from National Audubon Society, Frezza and Lorenz (unpublished).
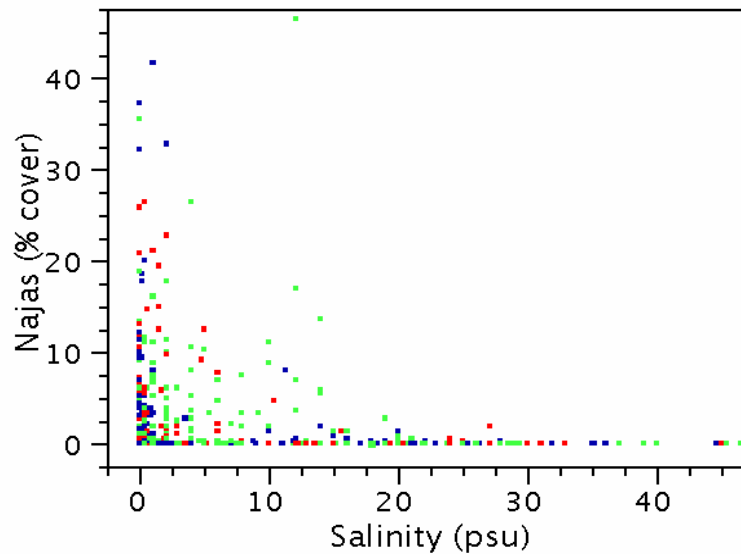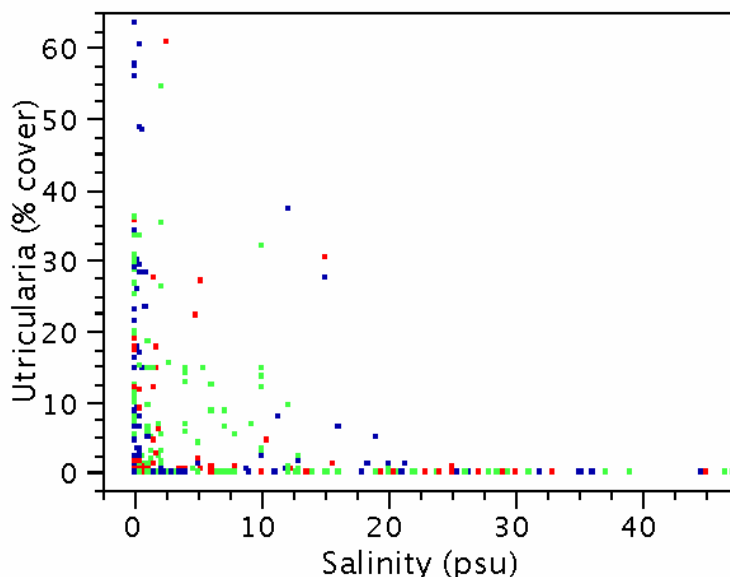

Figure 37. *Najas* (Waternymph) Cover in Relation to Instantaneous Salinity in the Transition Zone. Blue = Taylor River transect; red = Joe Bay transect; green = Highway Creek transect. Data from National Audubon Society, Frezza and Lorenz (unpublished).

Figure 38. *Utricularia* (Bladderwort) Cover in Relation to Instantaneous Salinity in the Transition Zone. Blue = Taylor River transect; red = Joe Bay transect; green = Highway Creek transect. Data from National Audubon Society, Frezza and Lorenz (unpublished).

SAV of the Transition Zone: Summary

Several lines of evidence from Florida Bay and other estuaries indicate that *Ruppia maritima* (widgeon grass) fills an important niche in the highly variable oligohaline-mesohaline region of estuaries. Throughout Florida, the southeastern United States and the U.S. Gulf coast, *Ruppia maritima* populations inhabit areas in which salinity ranges from 0 to about 30 psu. Inhibition of population growth at salinity levels above 30 psu is evident from Florida Bay surveys that found decreased areal coverage by *Ruppia* during periods of drought and elevated salinity. Preliminary statistical analysis of *Ruppia* distribution indicates an upper salinity limit of 25 or 30 psu for viable populations. While laboratory studies have found that *Ruppia* can tolerate high hypersalinity when other environmental factors are favorable, the absence of *Ruppia* at field sites under hypersaline conditions may indicate the importance of the interaction of salinity with other factors.

Hypersalinity may cause the long-term (mullet-year) loss of *Ruppia* from transition zone sites because of reproductive failure. Laboratory study of seed germination indicated that no recruitment occurs above 30 psu. Additionally, field observations were that no flowering occurs above 30 psu, pointing to a mechanism by which *Ruppia* populations are effectively confined to the mesohaline reach of estuarine systems. For Florida Bay, in order for local *Ruppia maritima* populations to reproduce successfully and be sustained, it appears that salinity must be below a threshold of about 30 psu when seeds are germinating and when seedlings are emerging.

Ruppia represents the best available candidate for an indicator species in the transition zone because of its ecologic importance and its role as the dominant rooted vascular macrophyte. The presence and condition of Ruppia at a site also provide an indication of pre-existing salinity conditions. In addition, the response of Ruppia to high salinity also closely tracks the responses of other important macroflora species that inhabit the mangrove transition zone, including *Chara* (muskgrass), *Najas* (waternymph) and *Utricularia* (bladderwort). *Ruppia* and *Chara* distribution and cover are fairly sensitive to salinity, greatly decreasing between 25 psu and 30 psu. *Utricularia* and *Najas* are significantly less salinity tolerant, with salinity thresholds at around 15 psu. This relationship places *Ruppia* at the upper limit of salinity tolerance for maintaining the low-

salinity macrophyte assemblage and SAV habitat of the transition zone. When *Ruppia* is impaired by excessive salinity, the freshwater/mesohaline macroalgae consortium is most likely already impaired or eliminated. If *Ruppia* is eliminated because of high salinity, the entire vegetation association characteristic of the transition zone, along with its habitat function, is probably also gone. Disappearance of *Ruppia* and the associated algal species from the northern coastal bays would be harmful to the low-salinity fauna of the transition zone that depend on this vegetation assemblage and other low-salinity SAV for food and habitat.

Northeastern Florida Bay and Its Submerged Aquatic Vegetation

A defining feature of the northeastern zone of Florida Bay, as of the bay as a whole, is its shallowness: the water of the northeastern zone averages about 1 meter in depth (Schomer and Drew 1982). As a result, sunlight sufficient to support photosynthesis can reach the sediment surface in almost all parts of the northeastern bay, resulting in dominance of seagrass beds as both a habitat and a source of primary production. This shallowness, combined with meager water exchange between the northeastern zone and the Atlantic Ocean or Gulf of Mexico (because of central and western bay mud banks), results also in long residence times and the potential for hypersalinity during droughts, as described in the earlier section on bay hydrology. Another defining feature of the northeastern bay is that phosphorus concentrations are extremely low: primary productivity is strongly phosphorus limited (Boyer et al. 1997, Childers et al. 2005).

The foundation of the Florida Bay ecosystem is its seagrass community (Zieman et al. 1989, Fourqurean and Robblee 1999, Rudnick et al. 2005). In the northeastern bay, both *Halodule wrightii* (shoal grass) and *Thalassia testudinum* (turtle grass) are common, with the *Halodule* being more common in less saline (often mesohaline–polyhaline) waters near the northern shoreline. Seagrasses are a highly productive foundation of the food web, a principal habitat for higher trophic levels and a controller of water quality, which they affect through 1) nutrient uptake and storage, 2) trapping of particles (within their leaf canopy) and 3) binding of sediments (with their roots). With growth of dense seagrass beds, these three water quality control mechanisms drive the bay toward a condition of clear water, with low nutrient availability for phytoplankton growth and low concentrations of suspended sediment in the water. Nearshore regions of northeastern Florida Bay, such as Little Madeira Bay) tend to have dense seagrass beds, but much of the remaining northeastern bay has relatively shallow sediments (depth to the limestone base), low phosphorus availability, relatively sparse seagrass coverage (compared with central and western Florida Bay) and high turbidity from suspended sediments (Stumpf et al. 1999).

Seagrasses provide refuge, spawning or nursery area and a food source for numerous important fish and invertebrate species (Zieman 1982, Sogard et al. 1989, McIvor et al. 1994, Thayer et al. 1999, Heck et al. 2003). Faunal growth, survival and abundance tend to be greater in the seagrass beds than outside the beds (Heck et al. 2003). Spotted sea trout (*Cynoscion nebulosus*), grey snapper (*Lutjanus griseus*), red drum (*Sciaenops ocellatus*), snook (*Centropomus undecimalis*), striped mullet (*Mugil cephalus*), bay anchovy (*Anchoa mitchelli*) and a variety of forage fishes are permanently or transiently resident in Florida Bay (Sogard et al. 1989, Johnson et al. 2004). Pink shrimp (*Penaeus duorarum*) and the spiny lobster (*Panulirus argus*) use much of Florida Bay as a primary nursery ground (Browder et al. 1999, Butler et al. 1995). Shrimp develop in the bay, favoring seagrass habitat, before migrating to the Dry Tortugas (Ehrhardt and Legault 1999). Lobsters use the bay as juveniles before emigrating across the Keys to the Reef Tract offshore (Davis and Dodrill 1989).

Most of the taxa of the popular game species and forage base species described in Chapter 2 have been collected baywide, with the abundance of the individual species varying from zone to zone (in relative and absolute terms). The northeastern zone of the bay supports relatively low abundances, variously attributed to the zone's comparatively lower primary productivity, its

reduced circulation and tidal range, its geographic isolation (lack of marine connectivity for offshore spawners) and its increased variability of salinity (Montague and Ley 1993, Ley et al. 1999, Browder et al. 2002). Moreover, some surveys (as described by Ley et al. 1999, Matheson et al. 1999, Browder et al. 2002, Powell 2003 and Johnson et al. 2004) do not support the idea that the northeastern bay zone functions as a significant estuarine nursery for important game and commercial fisheries.

These conclusions require important caveats. For instance, over recent decades, the northeast zone has been the subject of fewer faunal surveys than have other zones of the bay (Tabb et al. 1962, Thayer and Chester 1989, Powell et al. 1989, Sogard et al. 1989a and 1989b, Matheson et al. 1999, Thayer et al. 1999). Furthermore, perhaps one of the most thorough fish surveys specific to the northeast zone (as described in Ley et al. 1999) was conducted during a historic drought when the entire bay experienced prolonged hypersaline concentrations. Work that has included the northeast zone (described and used by Johnson et al. 2005) has captured significant numbers of forage fishes, especially in the families Engraulidae (anchovies), Cyprinodontidae (killifish), Syngnathidae (pipefish), Gerreidae (mojarras) and Gobiidae (gobies). These smaller forage species compose a majority of the northeast system's fish abundance and biomass, and these same species are also broadly distributed across the bay—facts that facilitated these particular fishes' use in the forage model, which are discussed later in this document.

The mobility of many of Florida Bay's fishes also indicates the importance of understanding hydrologic and hydrodynamic connections between the Everglades watershed and all regions of the bay. A literature review by Johnson et al. (2004) emphasized that all five species examined (bay anchovy, snook, spotted sea trout, grey snapper and pink shrimp) would benefit from a reduction in the coverage, intensity and duration of hypersaline conditions in the bay, especially during the summer and late fall, when salinity-sensitive, post-larval life stages are most abundant. Prolonged hypersalinity is less common in the northeast region, but the insufficiency of hydrologic and hydrodynamic modeling tools inhibits understanding and prediction of how such conditions are established in adjacent interior portions of the bay. Moreover, the review by Johnson et al. (2004) reinforced the importance of examining habitat quality beyond salinity effects, especially in terms of SAV density and type.

Background and Evaluation of the Literature

The seagrass community is involved in nearly every habitat and every trophic and physico-chemical function of Florida Bay's ecology and plays an extremely important role throughout the bay's ecosystem (Stumpf et al. 1999, Matheson et al. 1999, Fourqurean et al. 2002, Ley and McIvor 2002). This seagrass community is extensive, with a range that comprises virtually the entire bay, making it "one of the largest seagrass resources on earth" (Zieman 1982). Florida Bay seagrasses have been subjected to perturbations that have altered their productivity and composition, leading to a catastrophic die-off in 1987 (Robblee et al. 1991); even today they continue to exhibit impairment (Hall et al. 1999, Durako et al. 2002).

Despite their importance, seagrasses in Florida Bay were not systematically monitored prior to the 1980s and only fragmented information exists regarding seagrass ecology and environmental conditions of Florida Bay prior to that time. Tabb et al. (1962) qualitatively described the extensive seagrass community in central and eastern Florida Bay as consisting of mixed stands of *Thalassia testudinum* (turtle grass) and *Halodule wrightii* (shoal grass) or of dense monotypic stands of shoal grass.

Subsequent shifts in seagrass community structure in Florida Bay appear to have occurred in association with changes to bay hydrology and upstream landscape alterations for water management (Light and Dineen 1994) that were initiated in the early twentieth century and culminated in the 1960s. Historic information is rare, but Zieman (1982) and Zieman et al. (1999)

pieced together information about Florida Bay seagrass community distribution and succession from interviews with local watermen and unpublished reports. The bay was starved for fresh water for more than a decade during the 1970s, became a clear lagoon and was prone to episodes of hypersalinity. These conditions promoted the increasing dominance of turtle grass in both standing crop and spatial extent and decreased the prevalence of bare patches and shoal grass stands (Zieman 1982) throughout the bay.

Eastern Florida Bay was characterized by nutrient scarcity, thin and shallow sediments and the lowest overall abundance of *Thalassia* in the entire bay. A 1984 vegetation survey showed the eastern bay to be mainly comprised of sparse and patchy *Thalassia*, with a standing crop of 0–10 g dry weight m^{-2}, mixed with Halodule, which was more prominent in disturbed areas (Zieman et al. 1989). The tops of eastern bay banks often hosted denser stands of *Thalassia*, with a standing crop of up to 30 g dry weight m^{-2}. The leaves in these stands were often covered with epiphytes. Species of macroalgae of genera such as *Laurencia, Batophora, Acetabularia* and *Penicillus* were found in specialized eastern bay habitats such as the lee sides of banks and bedrock outcroppings. Meanwhile, in the central bay, by comparison, dense monospecific stands of *Thalassia* were present, usually on the order of 50–60 g dry weight m^{-2}, but there was little evidence of *Halodule*. The densest stands of Thalassia occurred in western Florida Bay, forming extensive beds of 75–125 g dry weight m^{-2} and up to 400 g dry weight m^{-2} on some bank tops.

Hall et al. (1989) noted that *Halodule* was distributed throughout the entire bay, with highest short-shoot densities (> 1500 shoots m^{-2}) in the western bay and lowest densities (0–1 shoots m^{-2}) in the southern bay. The eastern bay had intermediate densities of *Halodule*, in the range of 0–500 shoots m^{-2}. Montague et al. (1989) characterized the eastern bay as sparsely vegetated overall (0–600 g dry weight m^{-2}), with *Thalassia* and the alga *Penicillus* in the most saline part of the eastern bay (mean of 31 psu), grading to *Halodule* in areas of intermediate salinity (mean of 21 psu) and to *Ruppia, Batophora* and *Chara* at Florida Bay–Everglades transition zone sites (mean of 15 psu).

Seagrass Die-Off and Recent Changes in Florida Bay

Changes in seagrass distribution and density occurred because of a massive *Thalassia* die-off event that began in the fall of 1987 (Robblee et al. 1991, Hall et al. 1999). Die-off was first noted in the north central bay (in Rankin Lake) and in the southeastern bay (Robblee et al. 1991). The die-off quickly spread to western bay basins (Johnson Key basin and Rabbit Key basin) and continued through 1989. This initial event severely affected the SAV community, killing 4000 hectares of *Thalassia* beds outright and thinning the population in 23,000 additional hectares. Within the major die-off areas, 95 percent of plants were killed and mortality eventually consumed 30 percent of the entire *Thalassia* community in Florida Bay (Hall et al. 1999, Durako et al. 2002). The common factor across die-off sites was the rapid, near-total death of dense stands of *Thalassia*. The central bay and western bay sites were most severely impacted. In the western bay, die-off was practically nonexistent. The northeastern bay and its less dense stands were not affected by the initial die-off, but the ecosystem-wide impacts may have had indirect implications for this area.

Halodule (shoal grass) and *Syringodium* (manatee grass) were not involved in the initial die-off event (Zieman et al. 1999, Hall et al. 1999), but after primary die-off subsided, a general, slower decline of the seagrass community began, which involved these two species. *Halodule* declined markedly in the years following the die-off from 1989 to 1994 (Durako et al. 2002). *Halodule* and *Syringodium* are thought to have been adversely affected by the secondary effects of the initial die-off, notably by an increase in water column turbidity that began in 1991. This "secondary die-off" may be evidence of the keystone role that *Thalassia* plays in the survival of other benthic flora, as the light penetration characteristics of bay waters seem to have been altered by the loss of *Thalassia*'s sediment stabilization and nutrient uptake properties.

Seagrass cover continues to change dynamically. **Figure 39** provides change maps for *Halodule* and *Thalassia* cover in the 1995–2001 and 1995–2003 periods. The circled area is the part of the northeastern bay covered by this study, located along the gradient through the transition zone, includes Madeira Bay (west of Little Madeira Bay) and includes Eagle Key Basin (south of Little Madeira Bay). Results showed little change (gray shading) in *Halodule* in this region even as significant losses of *Thalassia* occurred there between 1995 and 2001, followed by a strong rebound of *Thalassia* between 2001 and 2003. Meanwhile, in the western bay, long-term losses in *Thalassia* continued, but increases in *Halodule* were observed.
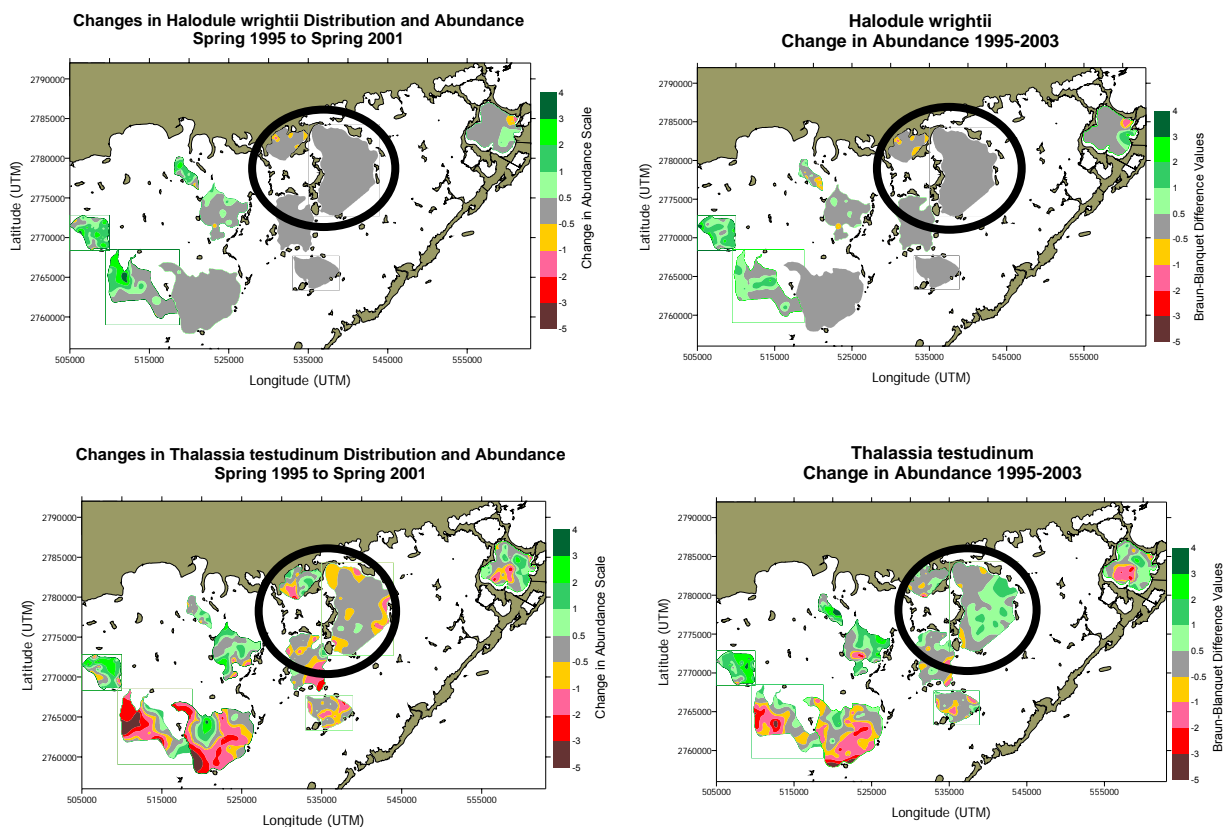
Figure 39. Changes in Florida Bay SAV Cover 1995–2003. Maps show areas of seagrass Braun-Blanquet density 1995–2001 and 1995–2003 for *Halodule* (upper) and *Thalassia* (lower); green tones represent areas of increasing SAV cover over the specified time interval, while red tones represent areas of decreasing SAV cover over these intervals. Circles indicate general region represented by the Everglades-Florida Bay Transition zone transect. Data from FHAP monitoring program, Durako and Hall (unpublished).

Salinity Responses of *Thalassia* and *Halodule*

Analyses of Field Data

Several studies have assessed the salinity tolerance and ranges of *Thalassia* and *Halodule* (reviewed in Battelle 2004). Mesocosm and field measurements indicate that the optimum salinity range for these marine plants is near full-strength seawater. Hanlon and Voss (1975) describe the optimum salinity range for *Thalassia* as 25–38 psu and note the plants' ability to tolerate

extremes of 11–48 psu; *Halodule* is noted as tolerant of salinity between 1–60 psu. Zieman (1982) describes the salinity range of *Thalassia* as 28–45, with maximum productivity at 35 psu.

In Florida's Caloosahatchee estuary system, Doering et al. (2002) found densest *Halodule* (>1500 m^{-2}) at salinities above 20 psu. Montague and Ley (1993) found *Thalassia* in Florida Bay at 20–40 psu. In Texas, Jewett-Smith (1991) measured high shoot densities of *Halodule* (5800–15,800 m^{-2}) in Redfish Bay, Nueces Bay and in the hypersaline Laguna Madre (40 psu). Dunton (1996) found healthy populations of *Halodule* in Laguna Madre at salinities up to 55 psu.

Tabb et al. (1962) observed in Florida Bay that *Thalassia* was found two plant forms: a thin and small profile plant in eastern bay areas subjected to widely fluctuating salinities (25–45 psu) and taller and more robust plant in areas of stable marine salinity. During a drought period ending in 1957, the short population along with a macroalgae assemblage dominated by *Caulerpa* increased in density. When the drought ended and salinity declined in the eastern bay, *Thalassia* declined in size and density until another drought began in 1961. *Thalassia* reached peak biomass during the second drought year of 1962. Tabb et al. (1962) observed plant die-back in Florida Bay at 45 psu, resulting in bare sediment substrate after 3–5 months.

Seagrass cover, shoot densities of *Thalassia* and *Halodule* were measured in northeast Florida Bay (Little Madeira Bay, Joe Bay and Long Sound) from April 1999 to September 2004 (Miami-Dade Department of Environmental Resource Management- DERM). Densities were compared to instantaneous salinity (**Figure 40**). These salinity data were too sparse to extrapolate 30-day means of salinity prior to sampling. In addition, the highest salinity value observed during the five-year period was about 43 psu, so the effects of more extreme hypersalinity that can occur in the region could not be inferred. The *Halodule* and *Thalassia* density measurements appeared to be independent of the instantaneous salinity measurements, meaning that it is difficult to assign a threshold salinity within this observed range that would be injurious to either species.

When short-shoot data are plotted against water depth (**Figure 41**) the two species have different distributions, with the mode for *Thalassia* around 0.8 m and for *Halodule* around 1.1 m. Even keeping in mind that *Thalassia* is a taller plant, based on these field observations, it appears that *Thalassia* density increases relative to *Halodule* in shallower water. It is difficult to infer causality from this relationship, but one possibility is that colonization by *Thalassia*, which has a higher light requirement than *Halodule* (Zieman 1982), may be favored in shallower water, whereas Halodule colonization is favored in slightly deeper and more turbid waters (Kenworthy and Schwarzchild 1995). This and other additional subtle competitive factors make it difficult to differentiate between the two species' salinity tolerances from simple field data. Both tolerate salinity well, but other interrelated background factors are in operation and must be considered. The decline of much of the *Halodule* in the bay subsequent to the *Thalassia* die-off event made it clear that environmental conditions must be maintained within appropriate ranges in order to support the entire community of seagrass species. The complexities of habitat requirements, including salinity effects, can be adequately understood only through a dynamic multivariate simulation model of seagrass community ecology. This kind of tool provides a means to simultaneously analyze all factors, including hypersalinity that affect the resource
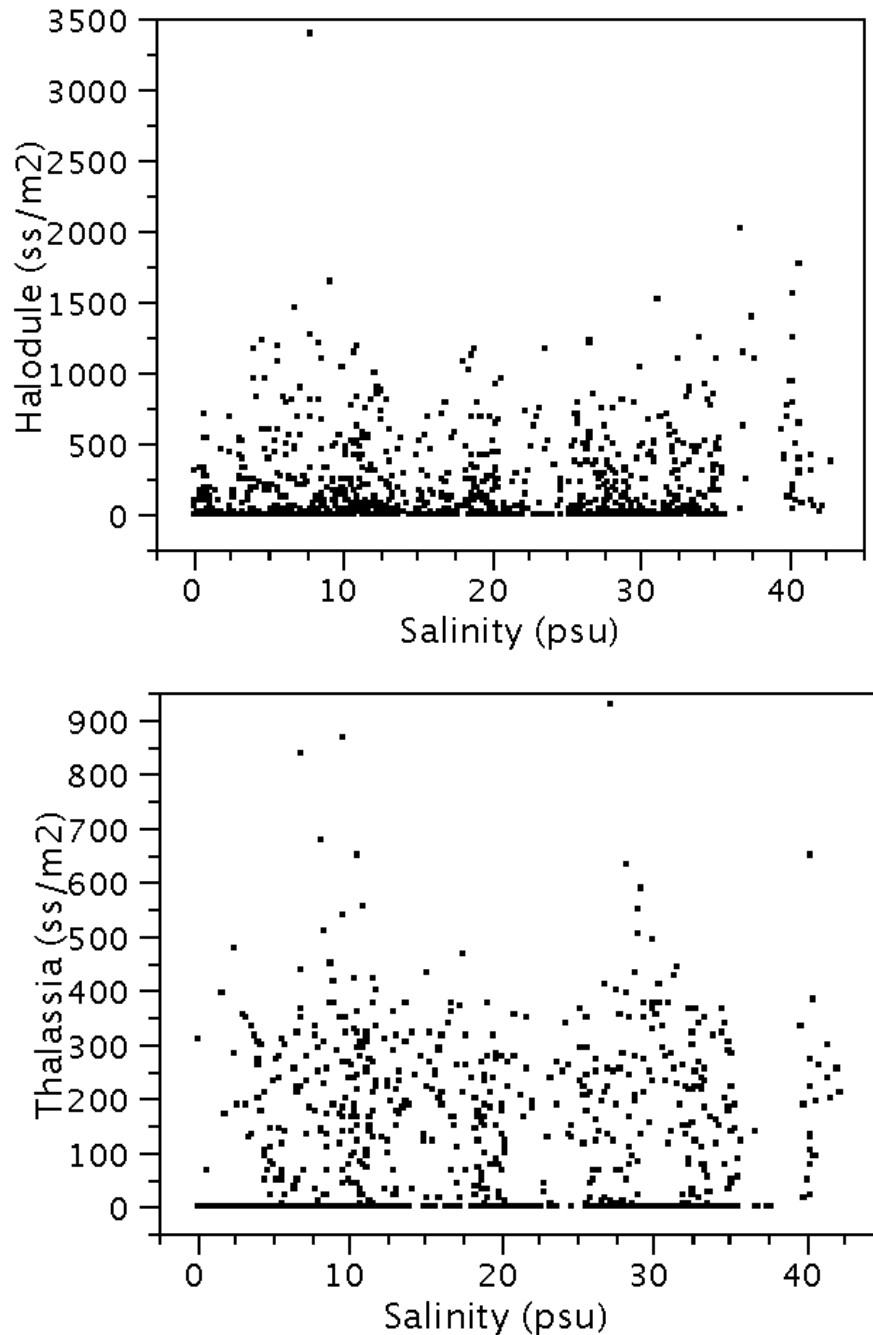
Figure 40. Density of *Halodule* and *Thalassia* Shoots in Relation to Salinity in Northeastern Florida Bay. Data collected by Miami-Dade Department of Environmental Resource Management (DERM) in Little Madeira Bay, Joe Bay and Long Sound from April 1999 to September 2004.
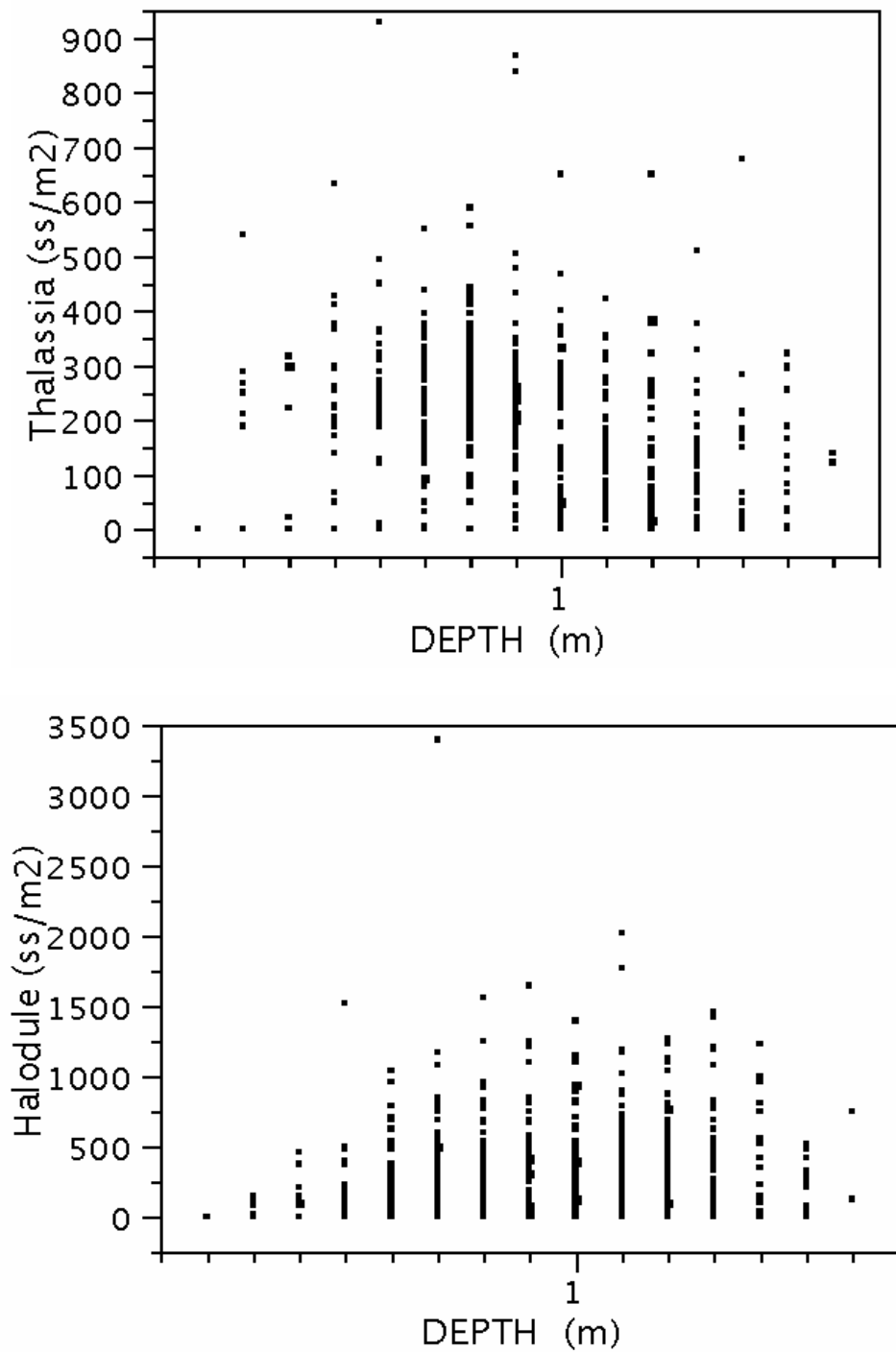
Figure 41. Density of *Thalassia* and *Halodule* shoots in relation to water depth in northeastern Florida Bay. Data were collected by Miami-Dade Department of Environmental Resource Management [DERM] in Little Madeira Bay, Joe Bay and Long Sound from April 1999 to September 2004.

Laboratory Analyses

Mesocosm studies were performed with *Thalassia testudinum* and *Halodule wrightii* plants from Florida Bay to determine their physiological tolerance to salinity and the optimal salinity range for the two species (Koch and Durako 2004). Results showed that in the short term, *Thalassia* survived and continued to produce shoot material at salinities as high as 50–60 psu. At 70 psu, plant standing stock biomass significantly declined. *Halodule* continued positive production of biomass up to 70 psu. However, in these experiments, measurements of osmolality and photosynthetic yield indicated that the energetic cost to the plant of maintaining an osmotic gradient, even at lower levels of hypersalinity, was significant and stressful to the plant. In the field, these species are rarely found at salinities as high as 60–70 psu, likely due to additional stress factors present in the environment and/or altered competitive capabilities.

These results indicate that high salinity alone is not sufficient to cause catastrophic losses of adult plants of *Thalassia, Halodule* or even *Ruppia*. As in the earlier described case with *Ruppia maritima*, moderately hypersaline conditions in controlled environments did not cause significant impacts on either *Thalassia* or *Halodule*, which were able to adjust internal solute concentrations osmotically to tolerate exposure to high salinities. Adult *Thalassia* and *Halodule* grown in sediments were tolerant of high salinity and maintained or even increased shoot numbers at salinities from 35 to 60 psu, when other physiochemical factors were held at optimal levels. In these experiments, plant standing stock began to decline significantly at 70 psu. Tissue osmolyte concentrations increased in all salinity treatments above 40 psu in both species, indicating plant stress and the energy expenditure needed to counteract the higher salinity outside the plant. Photosynthetic efficiency began to decrease at 60 psu in both species. The increase in osmolyte concentrations and decline in efficiency indexes indicate that the species were impaired in terms of energy balance and photosynthetic function. Shoot numbers were unaffected during the 60 day period that these plants were exposed to high salinities (Koch and Durako 2004). In a similar study with *Thalassia* seedlings, the young plants were found to be more sensitive than adult plants to high salinity and were unable to survive at levels above 50 psu (Koch and Durako 2004).

Although increased salinity alone is not sufficient to cause a catastrophic die-off, a combination of stress factors including salinity, higher temperature and higher sulfide can significantly decrease seagrass survival (Koch and Durako 2004). Leaf productivity rates and shoot counts of adult Florida Bay seagrass species did not change, or in some cases even increased, at salinities ranging from 35 through 60 psu (Koch and Durako 2004), but these same seagrass species were strongly impaired by moderate hypersalinity when an additional stress factor also was present. Mesocosm experiments by Koch and Durako (2004) tested the effects of combined salinity and temperature stressors on *Thalassia, Halodule* and *Ruppia*. With a small increase in temperature above average ambient bay temperatures, salinity of just 40 psu resulted in a 15 percent lower photosynthetic efficiency of *Thalassia* as compared with the efficiency at 35 psu. At 50 psu, efficiency was only 50 percent of that at 35 psu. When temperature was slightly elevated, *Ruppia* showed a similar but less pronounced response, exhibiting declines of about 15 percent efficiency at salinity of 40 and 20 percent at 50 psu. *Halodule* was not significantly influenced by temperature increases at any level of salinity. An earlier study of combined stressors (salinity plus sulfide) by Koch and Erskine (2001) showed that exposure of hydroponic *Thalassia* plants to a 6 μM sulfide concentration and salinity of 56 psu for two weeks resulted in a 50 percent decline in leaf biomass and shoot abundance relative to plants exposed to a similar sulfide level at 35 psu. Thus, high salinity alone may not provide sufficient stress to cause catastrophic die-off, but a combination of stress factors, including hypersalinity, temperature and sulfide can significantly decrease seagrass survival (Koch and Durako 2004).

Determination of an appropriate salinity range for sustaining a mixed *Thalassia testudinum* and *Halodule wrightii* assemblage in Florida Bay is a challenging task. The relationship of plant vigor to salinity in a controlled laboratory environment must be interpreted cautiously when extrapolating to the dynamics of populations in the field. Researchers are currently initiating

mesocosm experiments involving multiple species. Pending those results, computer models of the seagrass community are being used as a means to predict how elevated salinity will affect plant and community composition and survival in single and mixed species beds.

Ecological Modeling Analyses

A model developed for Florida Bay was used to examine responses of *Thalassia testudinum* and *Halodule wrightii* to multiple environmental stresses and to provide estimates of biomass under different freshwater flow conditions (Madden et al. 2003, Madden and McDonald 2006). The Florida Bay Seagrass Model is a set of separate spatially-averaged, mechanistic unit models calibrated to produce ecologic simulations of the seagrass community at different sites in different basins of Florida Bay. Response variables that are calculated include 1) species composition, 2) percentage cover and 3) biomass for each species. Important inputs to the model include salinity, inorganic nutrients, temperature, initial species composition, initial biomass, light and initial sediment sulfide concentrations. Other variables are internally derived, including concentrations of organic matter and interstitial hydrogen sulfide concentration in the sediments. The model runs with a three-hour time step using monthly mean estimated-salinity inputs from the FATHOM model for long simulation periods (33-year) and from averaged field salinity data for shorter (five-year) simulation runs. Salinity and temperature relationships are defined for each species on the basis of mesocosm studies described earlier (Koch and Durako 2004).

The dual-species model presented here incorporates the effects of interspecific competitive interactions between *Halodule* and *Thalassia.* The model is calibrated for two sites in the Taylor River's area of influence along the southern portion of the Everglades-Florida Bay Transition Zone transect (**Figure 23)** These sites are: Little Madeira Bay, near the mouth of Taylor River; and Eagle Key Basin, just south of the mouth of Little Madeira Bay. The baseline *Thalassia-Halodule* model uses averaged annual curves (interannual average for each Julian day) for the input variables from 1996–2001. The model produces stable populations of *Thalassia* and *Halodule* for both sites, and when calibrated with site-specific environmental data, the model simulates biomass for both target species agree well with field data for each site (**Figure 42**).

The model is used as a predictive tool, but care is taken in interpreting results because of the uncertainties in both the model itself and in the data used in the model. Model uncertainty has been examined (see Appendix I), and values of RMSE (root mean square error) for *Thalassia* biomass were 8.7 g-cm^{-2} in Little Madeira Bay and 3.1 g-cm^{-2} in Eagle Key Basin and for *Halodule* were 2.1 g-cm^{-2} in Little Madeira Bay and 1.1 g-cm^{-2} in Eagle Key Basin. Although some components are still in the parameterization process, the model represents the major processes and interactions in the seagrass community well. The results discussed in this study indicate that significant competitive interactions for nutrients and light occur among plants *in situ* and the outcome of this competition seems to be strongly influenced by salinity levels.
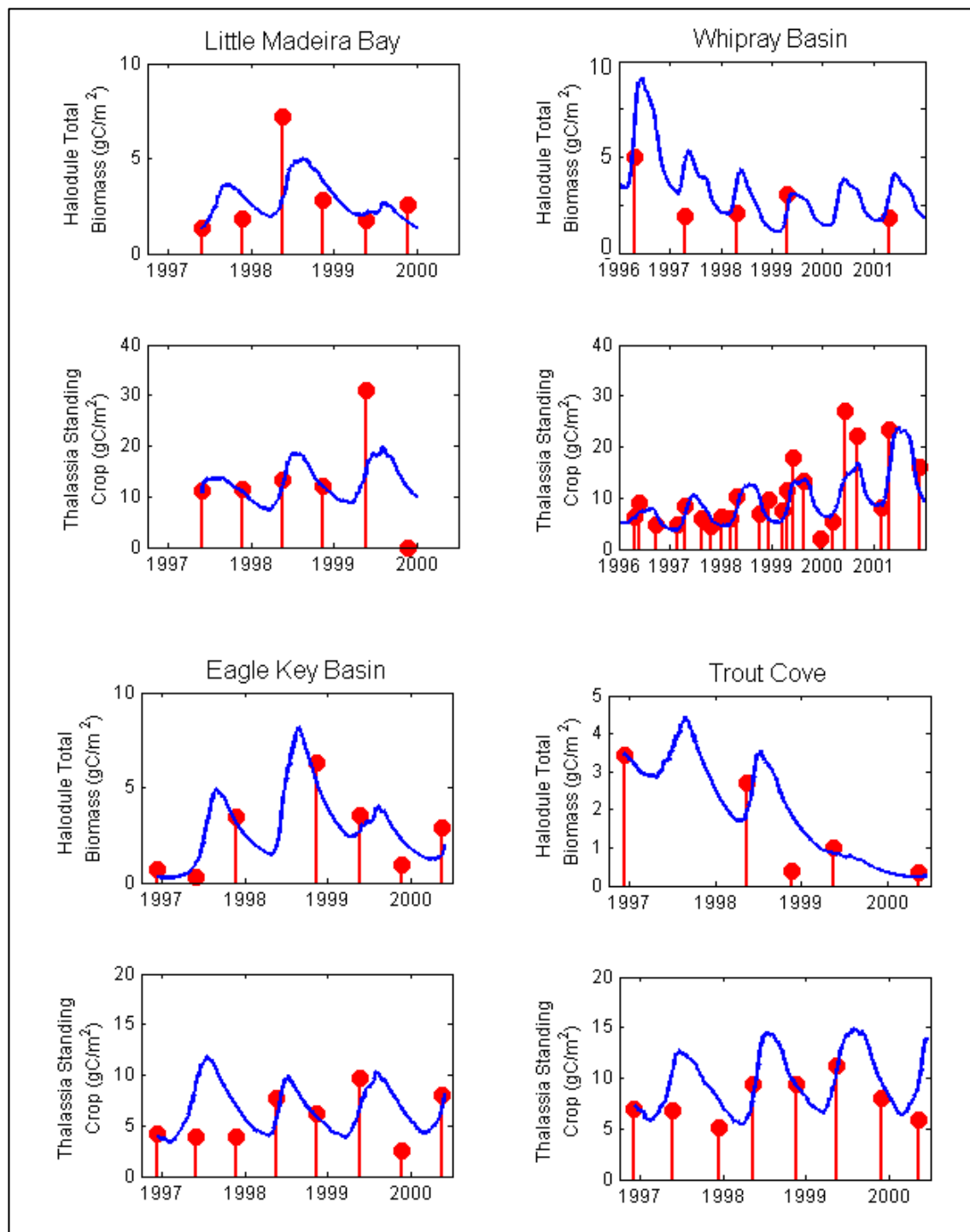
Figure 42. Calibration of SAV Biomass for Dual Species Seagrass Model for *Halodule* and *Thalassia* in the SAV model for Little Madeira Bay, Eagle Key Basin, Whipray Basin and Trout Cove. The latter two sites are not discussed in this MFL analysis. Model output [solid lines] for *Halodule* is total plant biomass, and for *Thalassia* is aboveground biomass, both in g-cm-2; solid circles represent calibration data from field measurements).

Use of the Model to Investigate Salinity Scenarios

The simulation model was used to investigate the effects of salinity variation on seagrass community dynamics. Different salinity regimes, corresponding to a range of freshwater flow rates, were systematically applied to the average five-year salinity baseline pattern and the model was run to predict changes in seagrass biomass and composition by species. Five-year simulations for the 1996–2001 period were run to analyze the effects of elevated salinity on SAV bed mortality and recovery at the Little Madeira Bay site. Baseline salinity inputs were derived from average values at the mouth of the Taylor River. For a sensitivity simulation presented in **Figure 43** the salinity level for each day was increased by 20 psu above baseline salinity, yielding a maximum salinity of about 50 psu and a minimum salinity of about 20 psu. This range approximately equals the levels observed near Little Madeira Bay during severe drought years (such as 1989 to 1990).
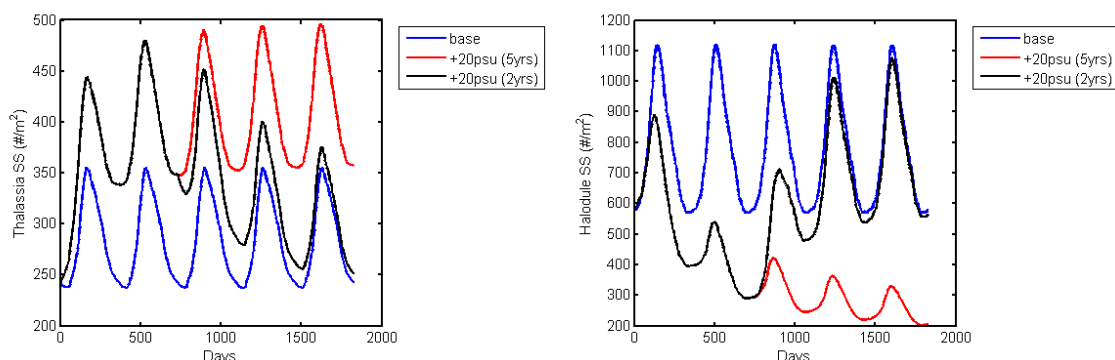

Figure 43. SAV model results from five-year simulations of elevated salinity in northeastern Florida Bay at the calibration location in Little Madeira Bay, along the Everglades-Florida Bay transition zone transect (see **Figure 23**). The figure shows *Thalassia* and *Halodule* short-shoot densities resulting from five-year simulations of the dual species model. Three treatments are shown; 1) a baseline salinity exposure, corresponding to average salinity in Little Madeira Bay repeated over five years; 2) an increase from the baseline by 20 psu for two years, followed by a return to baseline salinity for the next three years (black line); and 3) a five year period with salinity 20 psu above the baseline (black line through year two and red line thereafter). *Thalassia* responded favorably to elevated salinities, but *Halodule* rapidly declined and was impaired after two years that it did not immediately recover when salinities returned to normal. After a five year exposure to elevated salinity *Thalassia* remains elevated and *Halodule* essentially died off.

The salinity regime used in the model sensitivity analysis is similar to the intermediate salinity treatment in the mesocosm study by Koch (2003) and Koch and Durako (2004). Results from the model run showed that *Thalassia* had a strong positive response to increased salinity, and *Halodule* was impaired within two years and was eliminated within 5 years after increasing salinity by 20 psu (Madden and McDonald 2004). The response of *Halodule* modeled alone with elevated salinity was negligible (not shown); *Halodule* declined at higher salinity only in the presence of *Thalassia.*

The model was configured to allow a recovery from high salinity conditions. Salinity was reduced to baseline levels after two years in order to assess the SAV community's ability to recover to pre-stress levels. The early stages of the resulting five-year runs reflect the pattern just described for both species, with significantly lower *Halodule* biomass at higher salinities. *Thalassia* shows an increase of about 20 percent in biomass, which is maintained even after relaxation of the salinity stress after two years, reflecting a new equilibrium point for the population. In contrast, *Halodule*

responded weakly when salinity was relaxed to the baseline level, and recovery was only about 50 percent in one year and 90 percent in two years. *Halodule* did not return completely to the pre-stress biomass level even after three years of "normal" salinity. Indications are that when *Halodule* is impaired in this way for periods > 1 year, recovery times may be long because of increased dominance of *Thalassia* and because of mortality of *Halodule* seeds and belowground material and the ensuing low recruitment. This result of the model analysis supports field observations. *Halodule* virtually disappeared from its common range during 1989–1994 following the *Thalassia* die-off event and then remained persistently low for several years, despite a return to lower salinity conditions.

Long-Term Historical Retrospective Model Analysis

Another analysis was performed by modeling a 33-year (1970-2002) retrospective simulation of SAV trends using the calibrated SAV model and salinity estimates generated as output from the FATHOM model historical reconstruction. FATHOM salinity estimates for Basin 14 (Little Madeira Bay) and Basin 15 (Eagle Key Basin) along the Everglades-Florida Bay Transition Zone transect were used for the two simulation runs. This analysis was conducted to evaluate the effects of historical droughts and other low flow and high salinity conditions on the SAV community response over long periods and during periods when almost no environmental data or data on the SAV community were collected. The analysis enables us to provide a best estimate of seagrass community response to historically high salinity conditions.

Results from these model runs showed clear responses of seagrasses to salinity **(Figure 44)**, as *Thalassia* became the dominant species during periods of elevated salinity. During the three periods in the historical record when salinity remained above 40 psu for two or more consecutive years at the Little Madeira site, *Thalassia* growth was favored at the expense of *Halodule*. Immediately following extended periods of elevated salinity, increased freshwater flow from Taylor River resulted in lowered salinities. By the late 1990s *Thalassia* was nearly eliminated from the Little Madeira Bay site. In Eagle Key Basin, about 5 km from Taylor River mouth, salinity remained higher, favoring *Thalassia* and suppressing *Halodule* growth from 1970–2003. Briefly during the mid-1980s, and then persistently beginning in the mid-1990s, the onset of reduced salinities corresponded with increased *Halodule* biomass at Eagle Key and resulted in the development of a mixed *Thalassia-Halodule* assemblage.

The modeling results reflect changes in species composition for the *Thalassia -Halodule* community that correspond to field observations along the Everglades-Florida Bay Transition Zone transect (**Figure 23**) in Little Madeira Bay and Eagle Key Basin. This result differs from what might have been expected based on the outcome of the previously described mesocosm experiments, which indicated that *Halodule* and *Thalassia* have similar levels of salinity tolerance. The decline in *Halodule* as a function of salinity occurred in the model when both species were competing for the same resources. When modeled independently, the decline in *Halodule* biomass did not occur.
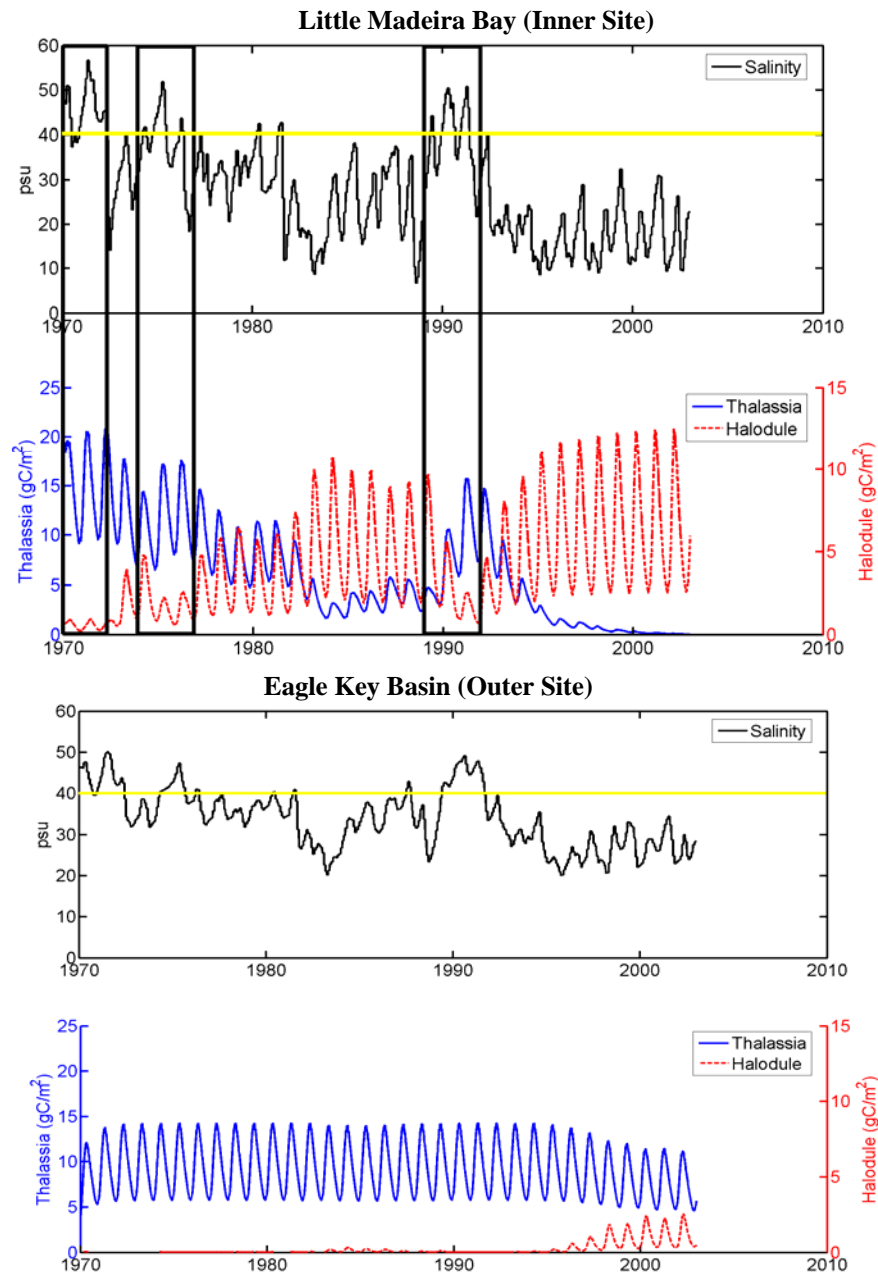
Figure 44. Seagrass Model Results from 1970–2002 Historic Reconstruction. The seagrass model was applied to [a] Little Madeira Bay [top panel] and [b] Eagle Key Basin [bottom panel] sites, located along the Everglades-Florida Bay Transition Zone transect (**Figure 23**). FATHOM predictions were used as input salinity; average monthly data were used for remaining environmental variables; the time series for salinity from the FATHOM model and biomass for *Thalassia* and *Halodule* are shown for the 33-year reconstructed historical period 1970–2002; three periods correspond to loss of *H. wrightii* at the inner site shown in the boxed area: [1] 1970–1971 drought, [2] mid-1970s and [3] 1989–1990 drought; in all cases, marine-to-hypersaline conditions prevail for >1 year; note development of monospecific *Thalassia* beds at the inner site in the early 1990s and decline in wetter years of the mid-1990s; at the outer site, *Thalassia* is the dominant seagrass from 1970 to the mid-1990s, when a mixed bed appears during the wetter period in the mid-1990s).

Mesocosm studies demonstrated that elevated salinities alone cause internal physiological stress (but not necessarily lack of growth) in seagrass plants even in otherwise ideal conditions of light, nutrients and oxygen. Similarly, the dynamics of interspecies competition may be shifted by high salinity *in situ* and in the model. Such a shift could cause *Thalassia* to outcompete *Halodule* for nutrients but also for light and space at particular locations. In the field, sulfide-rich sediments and interspecies competition appear to provoke a decline in the vigor of both species at elevated salinity levels (Madden et al. 2003). The model prediction derives from the reduced ability of *Halodule* to compensate for hypersalinity in the face of such multiple environmental stresses, which in turn reduces its ability to compete successfully with *Thalassia* for limited resources.

These model results reflect dramatic shifts that were actually observed in Little Madeira Bay and Eagle Key in recent years. It is instructive to look also at the longer-term field data on biomass for both species in Little Madeira Bay, which were collected from 1997–2003, as shown in **Figure 45.** Unfortunately, recent data beyond 2001 were not available for similar analysis of Eagle Key Basin.
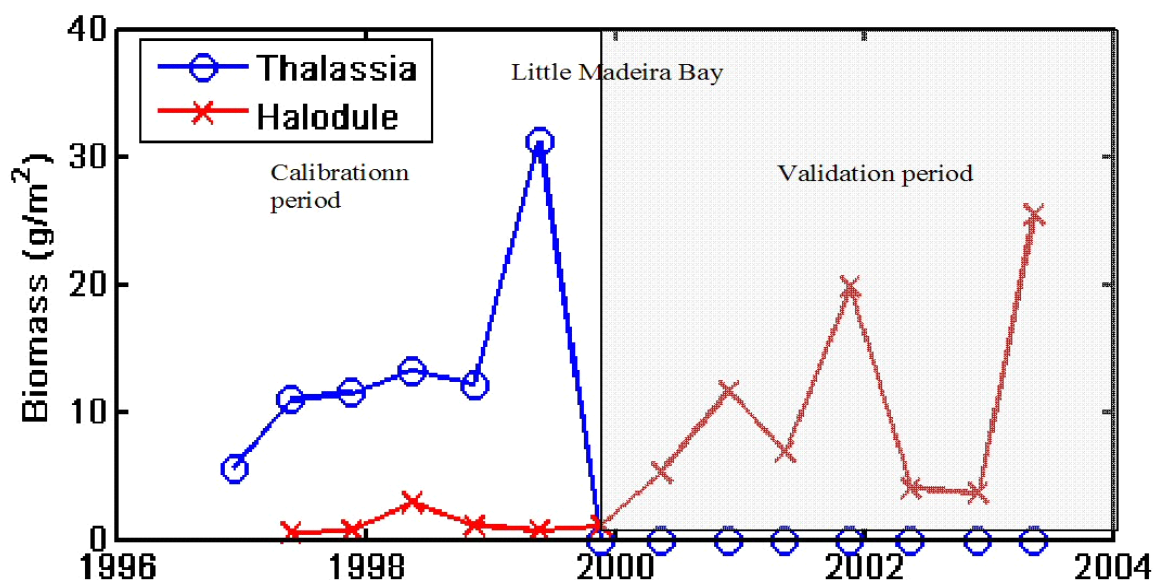

Figure 45. Measured Biomass of *Halodule* and *Thalassia* at the Little Madeira Site near the Mouth of Taylor River. The data include measurements used in the calibration of the seagrass model (1997–2001) and measurements in subsequent years (2001–2003), which can be used to validate the model. Compare this graph of observed data with model predictions shown at the top of **Figure 44.**

Though the 33-year model run did not incorporate these newer field data into its calibration dataset, the field response during the ensuing two years at Little Madeira Bay was consistent with model predictions. As flow increased in the late 1990s, *Thalassia* declined to zero and *Halodule* became the dominant species at the Little Madeira site. The species switch predicted for Little Madeira Bay toward the end of the 33-year model run (**Figure 44**) and the observation that such a switch actually occurred in the field after the model's calibration dataset time period, strengthens confidence in the model. The drop in biomass of *Thalassia* occurs earlier in the model than in the field and the population declines to zero over a period of years, whereas the drop in the field population was abrupt. Nonetheless, the model clearly responds to freshwater input (as predicted by the FATHOM model) and the disparity between data and model is likely a rate issue rather than a conceptual or structural issue in the model.

Output from the 33-year, dual-species, retrospective simulation was analyzed by regressing monthly SAV biomass outputs against monthly salinity estimate inputs. The result showed no

discernible trend for *Thalassia* biomass as a function of salinity (data not shown), supporting the conclusion that *Thalassia* is very tolerant of high salinity. In contrast, in a similar regression analysis of *Halodule*'s response to salinity in the dual-species model, biomass declined with salinities above 30 psu along the Everglades-Florida Bay Transition Zone transect at both the Little Madeira Bay and Eagle Key Basin sites (**Figure 46**). The illustration shows salinity ranges (light blue) in which *Halodule* achieved maximum standing crop; this area corresponds to salinities below 30 psu. The green areas reflect salinities (above 30 psu) at which *Halodule* biomass was much less than the maximum. The yellow areas of the graph show where *Halodule* failed to achieve significant biomass. As noted in earlier sections, these salinity responses (and especially the finding of distinct salinity thresholds) are not a direct result of the model's algorithm for photosynthesis and respiration as a function of salinity alone but are instead a secondary result associated with changes in interspecies competition as a function of salinity—an emergent property of the model.

The replacement of *Thalassia* by *Halodule* as predicted by the model for Little Madeira Bay reflects this emergent property. Both species individually tolerate a wide range of salinity, but competitive factors indicate that each species can gain an advantage within different portions of this range. The finding of greater *Halodule* salinity sensitivity in Eagle Key Basin versus Little Madeira Bay (**Figure 46**) may reflect an interaction between salinity and nutrient availability, since sedimentary phosphorus concentrations are lower in Eagle Key Basin than in Little Madeira Bay. *Thalassia* may have an advantage in nutrient uptake from the phosphorus-poor sediments because it has a deep and expansive belowground biomass and greater capacity for internal nutrient storage.

SAV of Northeastern Florida Bay: Summary

The seagrass community of northeast Florida Bay tolerates high salinity for limited periods, but other concurrent factors must also be taken into account when making decisions about freshwater input to the estuary. Mesocosm experiments and modeling analyses show that when salinity increases above historical levels, shifts in seagrass physiology and in the population dynamics of seagrasses make the community less stable and diverse. When conditions shift too far, a major impact, such as seagrass die-off, can ensue, resulting in a cascade of ecosystem effects. Several layers of analysis may be required to make the connection between physiology of a particular species and overall community or ecosystem dynamics. Available information and analysis is sufficient to warrant caution in allowing salinity levels to rise above 40 psu in northeastern Florida Bay, particularly at certain times of the year. Based on the model output, *Halodule wrightii* appears to be especially vulnerable to high salinity conditions during the late dry season and summer when temperatures are elevated and water circulation is restricted.

Mesocosm experiments did not show a strong sensitivity of individual plants to high salinity in the short term (weeks to months), but model analysis of hindcast conditions and historical field data on long-term trends indicates that elevated salinity affected the *Thalassia-Halodule* community complex, including effects on species composition, succession, and ecological function. When combined with other "natural" stresses such as nutrient limitation and sulfide toxicity, high salinities were shown by mesocosm studies and model projections to compromise seagrass community function by reducing *Halodule* cover and increasing *Thalassia*. In particular, when salinity in the model was persistently above 40 psu, species dominance in the *Thalassia-Halodule* assemblage shifted to favor *Thalassia* at the expense of *Halodule* and to drive the system toward an unstable monoculture.

Additional information must be gathered before making definitive determinations, but it is likely that salinity levels persistently above 40 psu are detrimental to the Florida Bay ecosystem.
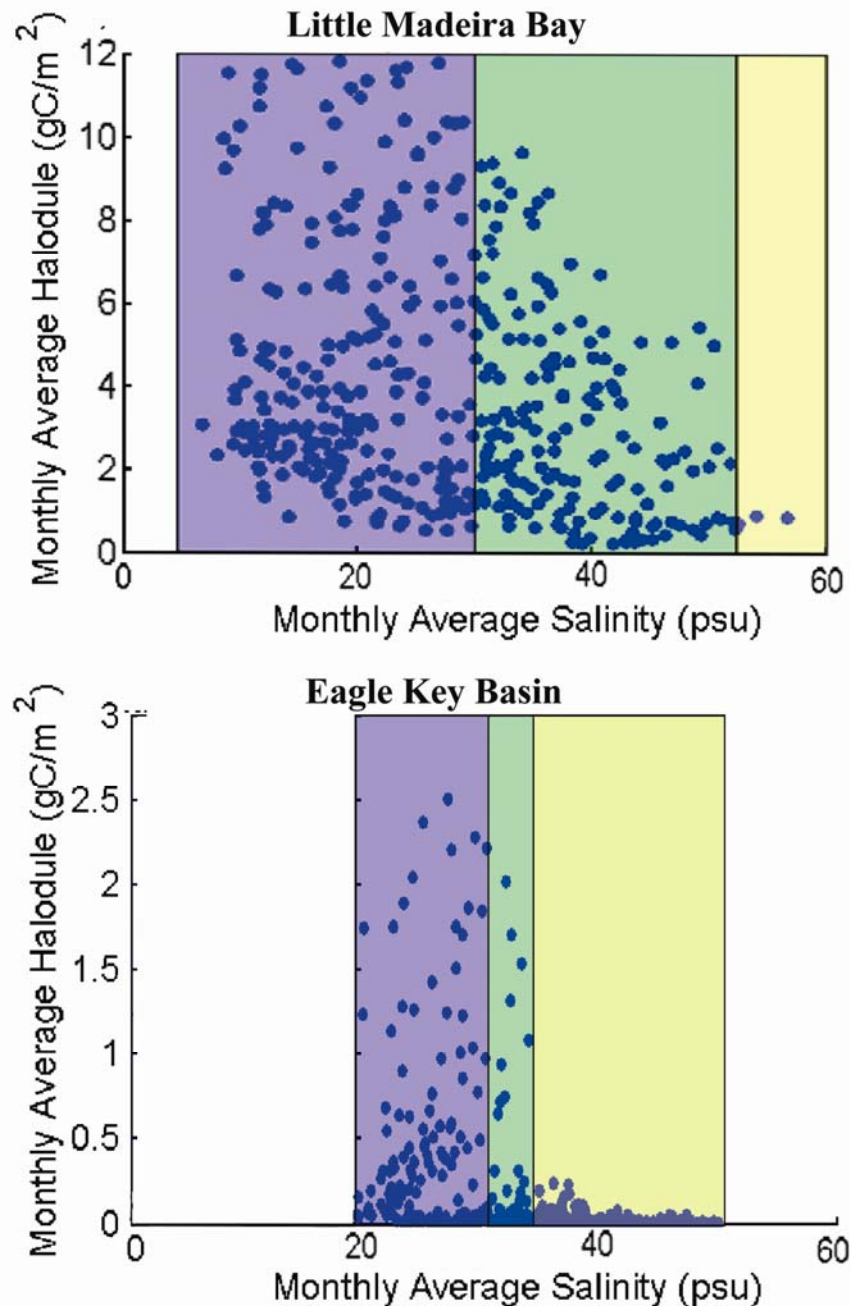
Figure 46. Seagrass Model Simulation of *Halodule* Biomass in Relation to Salinity from a 33-Year Historical Reconstruction. FATHOM salinities were used as input for two sites along the Everglades-Florida Bay Transition Zone transect (**Figure 23**). This analysis shows a strong decline in biomass at both inner Little Madeira Bay (top panel) and Eagle Key Basin (bottom panel) sites with increasing salinity. The blue area shows salinities at which *Halodule* achieves maximum standing crop (below 30 psu for both sites). The green area corresponds to salinities at which *Halodule* standing crop is declining. The yellow area shows where *Halodule* fails to achieve biomass values significantly above zero and is considered severely or lethally impaired). Note difference in vertical scales of the two plots, which reflect differences in overall productivity between these two regions.

Analysis of Higher-Trophic-Level Species of Northeastern Florida Bay

Statistical models were built to examine how Florida Bay fish and invertebrate species respond to variables of habitat and salinity. The General Additive Model (GAM) approach used for this work is a relatively recent development in statistical modeling that has been used in a number of ecologic and fishery population studies (Swartzman et al. 1992, Augustin et al. 1998, Fewster et al. 2000, Clarke et al. 2003, Ciannelli et al. 2004). Similar to the more common technique of multiple linear regression, GAMs relate the dependent variable to possibly important independent variables (covariates). Covariates in the GAM approach are assumed to affect the dependent variable through additive and independent unspecified (not necessarily linear) functions, thereby allowing changes in abundance to be related to covariates without restricting the functional form of the relationship (Hastie and Tibshirani 1990).

GAMs were built that take into account spatial and temporal variability in Florida Bay fisheries datasets from multiple studies spanning a sampling period of nearly three decades (mid-1970s through 2001). Models were developed for seventeen common Florida Bay forage species and three predator species (two that were represented in the database as juveniles). Faunal samples were collected with three types of gear: throw traps, seines and trawls. Sampling gear bias was considered by developing separate throw-trap models for some species and combined trawl and seine models for other species. Continuous independent variables in these models included salinity, depth, water temperature and SAV density or standing crop for each common SAV species -- *Thalassia testudinum, Halodule wrightii* and *Syringodium filiforme*. Categorical variables included habitat type, Julian day and sample region (as aggregations of FATHOM basins).

Each of the modeled species is mobile and was collected in samples from all regions of Florida Bay, although abundance per species varied regionally (which is why region was added as a variable). For each species, data from all samples (bay-wide) were used in model development. Additional model development details may be found in Johnson et al. (2005). Simplified visual examples of bivariate relationships between salinity (X-axis) and log-transformed density (Y-axis), holding all other variables constant, are shown in **Figure 47**.

These plots are simplified depictions of conditions actually experienced by the species in the field and represented in the multivariate models. They provide a useful summary of relationships among environmental variables and predictions for species density and trends. Because GAMs were used for this work, relationships between fish/invertebrate density and a variable were often complex. The mathematical structure of GAMs allows for univariate sensitivity analyses (for instance, as illustrated in **Figure 47** for salinity) holding other variables constant. As other variables (those not depicted on these plots) change in value, the slopes of the bivariate relationships for salinity will remain the same, while the intercept may change.
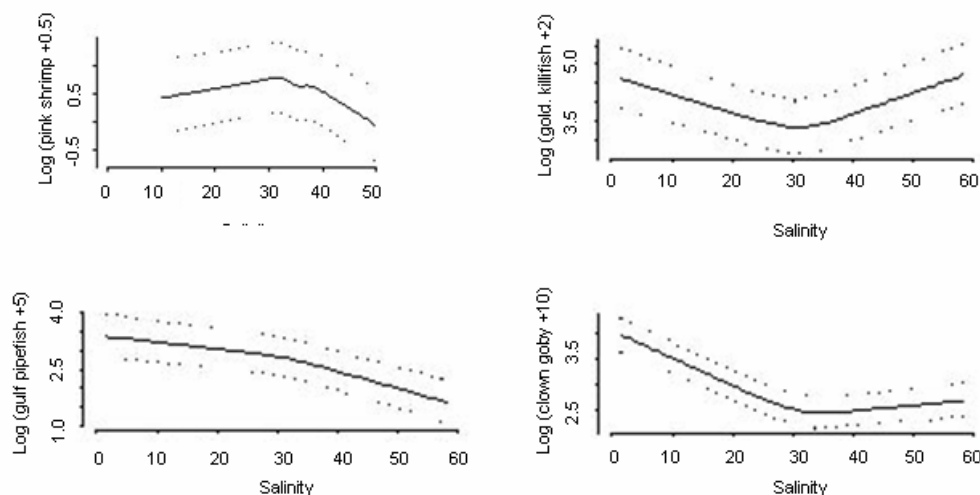
Figure 47. Statistical Model of Higher-Trophic-Level Species' Biomass as a Function of Salinity in Florida Bay. Graphs are examples of bivariate plots between log-transformed species' density and salinity, holding all other variables constant, with 95 percent confidence intervals (dotted lines).

Similar bivariate plots were interpreted to obtain the results presented in **Figure 48** for covariates salinity and density of *Thalassia, Halodule* and *Syringodium*. Salinity and at least one SAV species were significant variables in the GAM models for almost all taxa. Faunal density varied in a more complex manner as a function of salinity and *Thalassia* density than as a function of *Halodule* and *Syringodium* density. For the latter two SAV species, faunal density almost uniformly increased with increasing SAV density. For *Thalassia*, faunal density commonly increased with increasing plant density only when this SAV was sparse or moderate. Trends with salinity varied widely among species, but for most species, shifts in density occurred near marine salinity (30–35 psu).

A separate GAM analysis, using output from the dynamic Florida Bay Seagrass Model (described earlier), was performed to assess the interactive effects of salinity and SAV in northern Little Madeira Bay (near the mouth of Taylor River) (see **Figure 23**). A baseline salinity scenario was first developed by calculating average (by Julian day) salinity and temperature conditions from field data for this area. The seagrass model was then run for a five-year simulation period (as described in the SAV salinity scenario subsection, as for **Figure 43.** Model runs were repeated, using the average daily temperature data and different salinity values that were adjusted to represent increases in salinity relative to the average baseline values (by 5, 10, 15, and 20 psu). After SAV biomass was estimated by the Seagrass Model, these output data were converted to Braun-Blanquet Cover and Abundance (BBCA) density values (for each species) and input as monthly averages into the GAM trawl/seine models for each fish/invertebrate species. For each salinity scenario, associated inputs of salinity and temperature (also as monthly averages) that were used as input into the SAV model were also used for the GAMs. *Syringodium* cover was input to the GAMs as zero because this species is not found in the basin used for this analysis. Examples of inputs to the GAMs are shown in **Figure 49**.

Species	Salinity	*Thalassia*	*Halodule*	*Syringodium*
Thor sp. (caridean shrimp) [tt, f]	35 (increasing trend)	(increasing)	(increasing)	(increasing)
Hippolyte sp. (caridean shrimp) [tt, f]	(decreasing)	(increasing)	(increasing)	(increasing)
Farfantepenaeus duorarum (pink shrimp) [tt, t/s, f]	30 (peak)	tt (increasing)	tt (increasing)	(increasing)
Floridichthys carpio (goldspotted killifish) [tt, t/s, f]	30 (V-shape)	t/s (decreasing)	NS	t/s (decreasing)
Lucania parva (rainwater killifish) [tt, t/s, f]	t/s 30 (decreasing)	t/s (increasing)	t/s NS	(increasing)
Syngnathus scovelli (gulf pipefish) [tt, t/s, f]	(decreasing)	t/s (peak)	(increasing)	t/s (increasing)
Anarchopterus criniger (fringed pipefish) [t/s, f]	30 (decreasing)	(peak)	(increasing)	(increasing)
Syngnathus floridae (dusky pipefish) [t/s, f]	30 (peak)	(increasing)	(increasing)	(increasing)
Hippocampus zosterae (dwarf seahorse) [t/s, f]	30 (peak)	(peak)	(increasing)	(increasing)
Microgobius gulosus (clown goby) [t/s, f]	30 (decreasing)	(decreasing)	NS	NS
Microgobius microlepis (banner goby) [t/s, f]	(decreasing)	NS	(decreasing)	NS
Gobiosoma robustum (code goby) [tt, f]	40 (peak)	(increasing)	(increasing)	NS
Opisthonema oglinum (atlantic thread herring) [t/s, f]	NS	(decreasing)	NS	(decreasing)
Anchoa mitchilli (bay anchovy) [t/s, f]	35 (decreasing)	(decreasing)	(increasing)	(decreasing)
Atherinomorus stipes (hardhead silverside) [t/s, f]	(increasing)	(V-shape)	(increasing)	NS
Euchinostomus sp. (mojarras) [t/s, f]	30 (increasing)	(increasing)	(increasing)	(increasing)
Lagodon rhomboides (pinfish) [t/s, f]	30 (increasing)	(increasing)	(increasing)	(increasing)
Opsanus beta (gulf toadfish) [tt, t/s]	tt 30 (increasing)	tt (peak)	tt (increasing)	(increasing)
Juv. *Cynoscion nebulosus* (spotted seatrout) [t/s]	(decreasing)	(peak)	(increasing)	(increasing)
Juv. *Lutjanus griseus* (grey snapper) [t/s]	35 (increasing)	(increasing)	(increasing)	(increasing)

[tt] throw trap model   [t/s] trawl/seine model   [f] used as forage species for assemblage analysis

Figure 48. Summary of Results from Statistical Models for Twenty Florida Bay Fish and Invertebrate Species. The line plot within each cell indicates the generalized trend of species density as a function of the magnitude of a covariate, salinity (first column) or SAV density (by SAV species); "NS" designates variables that were not significant to a respective species model. Fourteen of these species (fish species indicated by "f" superscript) were selected as typical forage species for further impact assessment (see text).
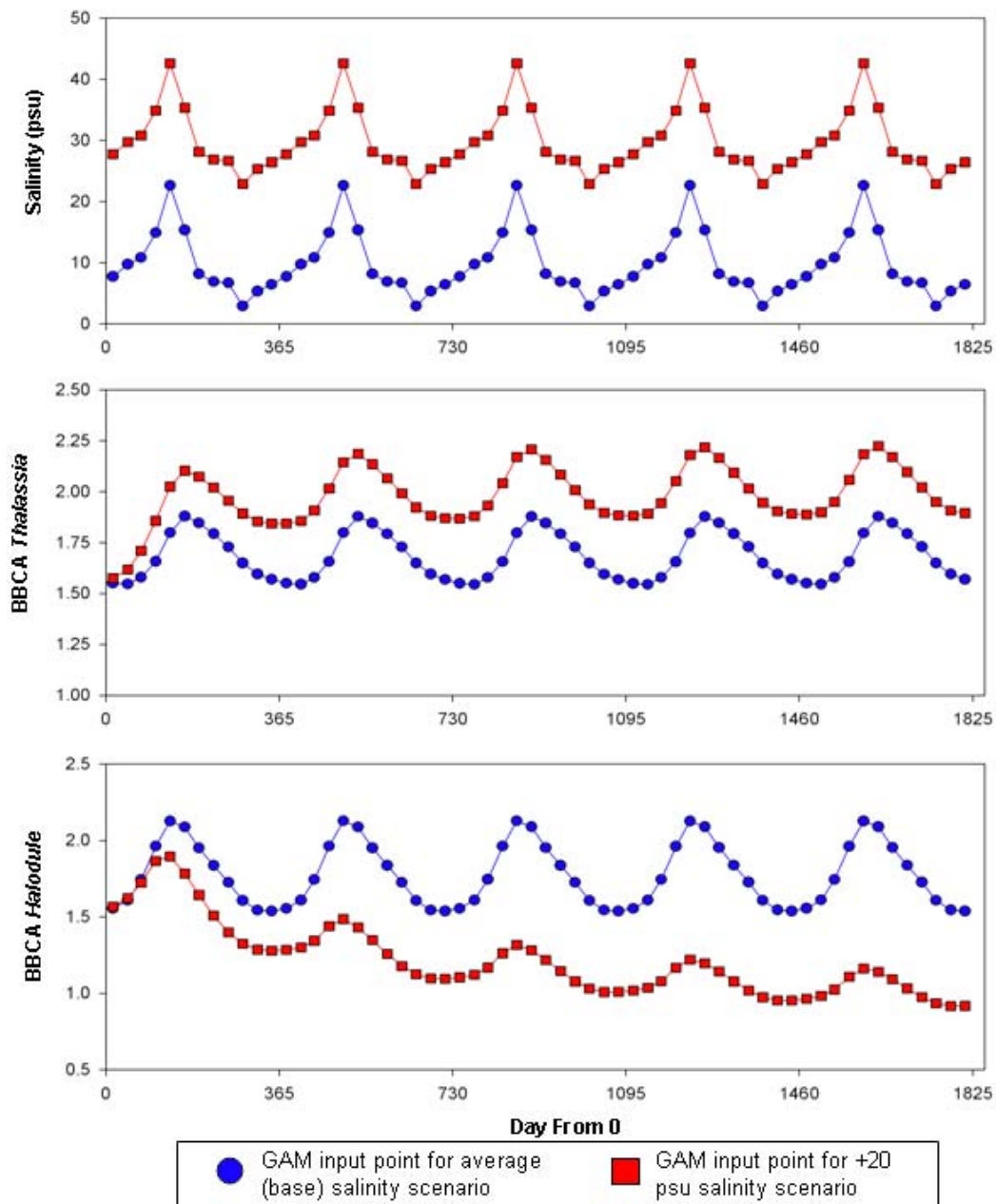
Figure 49. Salinity and SAV Habitat Inputs to the Higher-Trophic-Level Statistical Model, to Test Direct and Indirect Effects of Salinity Change on Animal Species. Plots show monthly salinity and seagrass cover (Braun-Blanquet Cover Abundance, BBCA) input data to GAMs, used to test salinity sensitivity of the fish models; above are time series results for two (of the five) salinity scenarios, run over five years through the Florida Bay Seagrass Model for the Inner Little Madeira Bay basin. These inputs to the GAMs were monthly averages of the Seagrass Model's daily input (salinity and temperature) and output (biomass of each SAV species, converted to a BBCA value [0–5 scale]).

The forage fish GAMs were used to make predictions for each month of the 5-year SAV model simulation. Results from the scenarios were examined for each fish/invertebrate species and were aggregated to assess effects on a forage assemblage composed of the 14 species identified in **Figure 48**. These assemblage results are shown in **Figure 50** for the average (base) salinity and 20 psu boosted salinity scenarios across the simulation period. The lower panel of **Figure 50** also shows predicted forage fish densities for the 20 psu boosted scenario as a proportion of the predicted densities for the base salinity scenario. While these results are presented for ease of interpretation as a time-series, it is important to note that the GAMs are independent of time and represent only static ("snapshot") results. Thus, the results should be viewed as conservative: these models do not reflect dynamic effects of predator/prey interactions or competition, population recruitment or other life history traits over time. This is in contrast to the SAV model that is capable of incorporating dynamic feedbacks into its results as time series outputs.

One noteworthy trend in the results from **Figure 50** is a consistent depression in predicted fish abundance in the higher salinity scenario versus the base case salinity. Though there is some month-to-month variability, the average decline in predicted forage fish density caused solely by the 20 psu increased salinity was approximately 15 percent. This trend in assemblage-level results was driven in large part by a subset of dominant species that declined based on salinity alone (e.g., bay anchovy and the 2 killifish species). By the end of the 5-year simulation period, the proportion of fish predicted to occur in the high salinity scenario dropped to (on average) just below 70 percent of that predicted for the base salinity condition. Because the annual salinity and temperature curves were repeated across the simulation period, this additional decline (of over 15%) can only be explained as a result of a change to the SAV inputs, specifically the drop in *Halodule* over time. As shown in **Figure 49**, there were several forage species that were predicted to decline as density of this SAV species dropped. One dominant member of the forage assemblage, *Euchinostomus* sp. (mojarras), was more sensitive to declining *Halodule* density than to the higher salinity concentrations in the raised salinity scenarios.

The importance of SAV habitat to the forage base assemblage is also evident in **Figure 51** in which the predicted forage fish densities for *all* salinity scenarios (and corresponding Seagrass Model outputs) are plotted versus both salinity and *Halodule* inputs for the month of April (for all years of the simulation period). In this plot predicted fish density drops in conjunction with both increasing salinity concentrations and decreasing density of *Halodule*, validating the importance of predicting the effects of these habitat conditions in concert. The results suggest that salinity effects on fauna occur not only directly via physiological stress on the fauna but also via habitat modification. This coordinated modeling exercise demonstrated 1) the importance of salinity and habitat as interactive factors that influence Florida Bay's higher trophic level species and 2) the validity of using SAV habitat as an ecosystem indicator for MFL development.
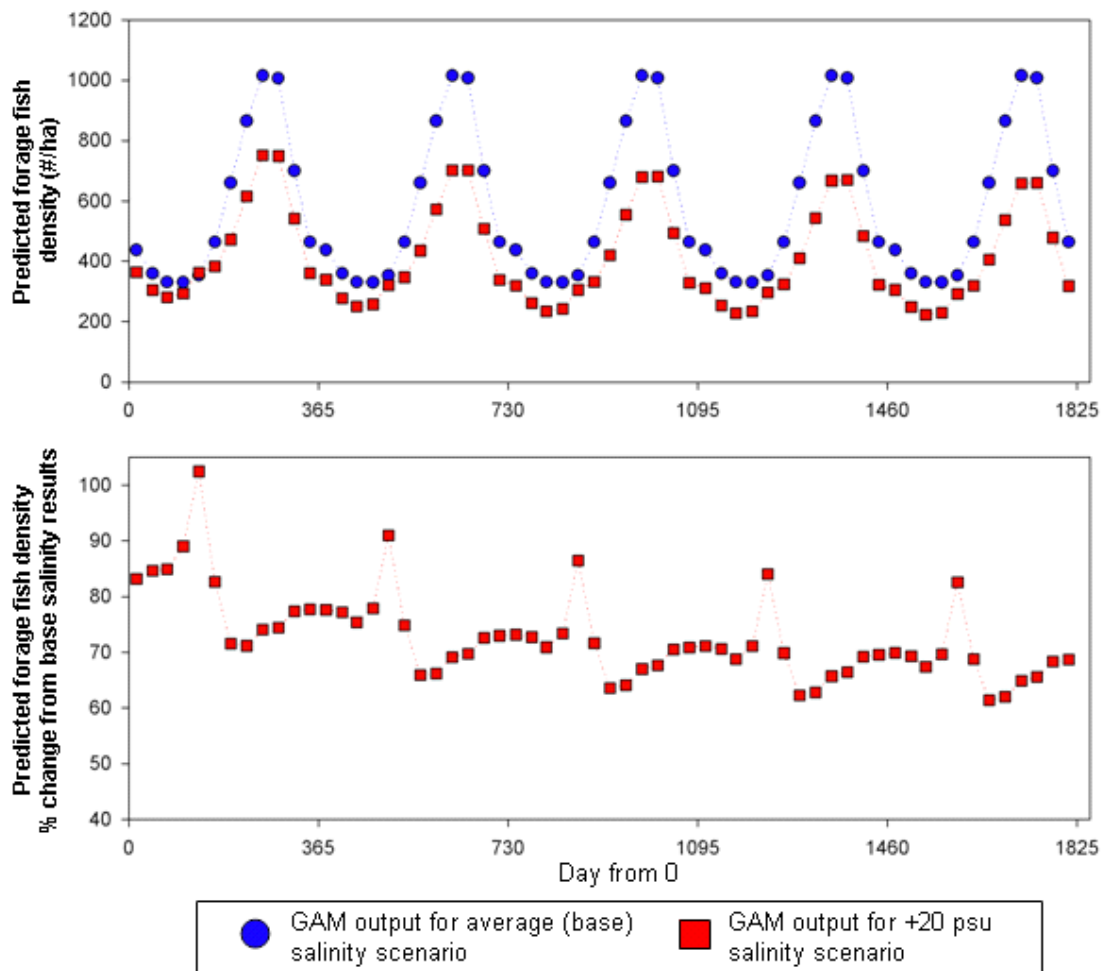
Figure 50. Results from Higher-Trophic-Level Statistical Models of an Assemblage of Fourteen Species, Showing Direct and Indirect (via SAV Habitat Change) Salinity Effects Over 5-Year Simulation Period. Results are estimated by GAM statistical models (trawl gear type only) for two (of the five) salinity scenarios run for the Inner Little Madeira Bay basin. Results are displayed as time series output for ease of interpretation with input datasets (**Figure 49**), though the GAMs themselves are static models that do not incorporate results into subsequent simulations over time. The top panel shows predicted fish densities (as a summation of prediction densities for the 14 modeled forage species) for both base salinity and 20 psu boosted salinity scenarios. The bottom panel displays the results for the 20 psu boosted salinity scenario as a proportion of the results for the base salinity scenario, and specifically highlights the effect of declining Halodule density over the simulation period in which the annual salinity curves were repeated.
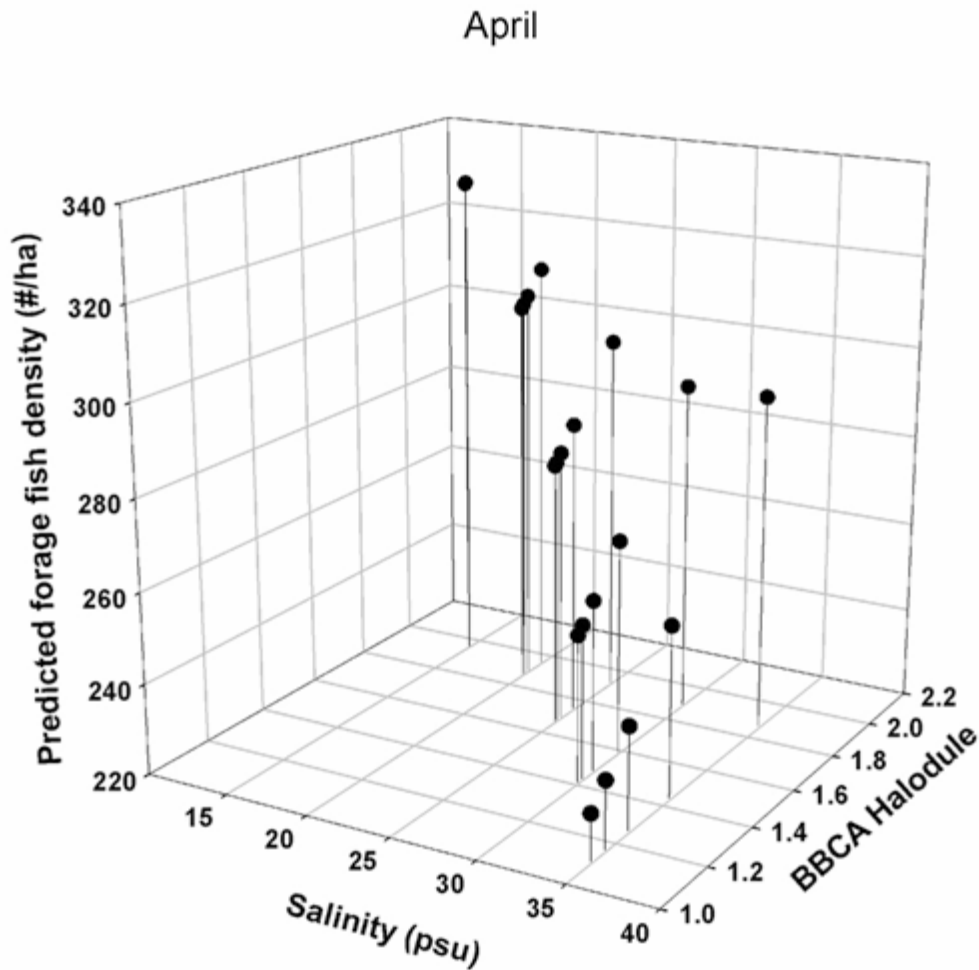
Figure 51. Results from Higher-Trophic-Level Statistical Models of an Assemblage of Fourteen Species, Showing Concurrent Effects of Salinity and SAV Habitat Change for All Salinity Scenarios. Results are estimated by GAM statistical models (trawl gear type only) for the Inner Little Madeira Bay basin for the month of April. For each salinity concentration, any change in BBCA for *Halodule* represents a change over the simulation period (note that for each scenario, salinity was held constant amongst years for each month). Changes in *Halodule* density over time had a compound effect with salinity on the predicted fish density.